A spatially distributed ammonia emissions inventory for the UK

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Declaration

This thesis has been composed by myself, and all work reported herein is my own except where otherwise stated.

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Abstract

Ammonia (NH_3) emissions originate mainly from agricultural sources and provide a major contribution to the eutrophication of nitrogen sensitive ecosystems as well as the acidification of soils and water bodies. Accurate spatially distributed emission estimates at both the national and the local scale are an essential input to models of atmospheric transport, deposition and critical loads exceedance. The distribution of NH_3 air concentrations and deposition over the UK is characterised by a high spatial variability. Previous studies have highlighted that this is due to a) the high spatial variability in the distribution of NH_3 sources over the country, and b) the fact that NH_3 is highly reactive with a large proportion being deposited close to the sources.

In this study, a new methodology for a spatial NH_3 emissions inventory for the UK has been developed. In contrast to previous methodologies, the new approach employs a spatial model specifically tailored to NH_3, rather than a more general allocation of agricultural sources (livestock and crop categories). This is important as NH_3 emissions do not occur equally over all the possible spatial locations of the sources. Key model input data are agricultural census data (updated to 1996), average nitrogen fertiliser application rates to crops and grassland, landcover data and NH_3 source strength estimates. Component sources such as livestock grazing or manure spreading are weighted by the magnitude of their emission source strength and distributed onto suitable landcover types at a 1 km grid level. At present the model results are aggregated to a 5 km resolution to reduce uncertainty in the spatial location of NH_3 sources. The inclusion of Northern Ireland into the inventory and the new spatially distributed estimates of non-agricultural NH_3 sources modelled in this thesis resulted in the most comprehensive spatial NH_3 emissions inventory for the UK to date.

A comparison of the results with previous models showed that emission estimates have been decreased to more realistic levels in extensively used upland and hill areas, and concentrated in intensively used agricultural areas. This has major implications for the estimation of critical loads exceedances for intensive lowland versus extensive upland areas. Both the overall magnitude and the spatial distribution of NH_3 emissions presented in this thesis are strongly supported by a comparison of air concentration fields derived from the new model with the results of the National Ammonia Monitoring Network. This has been shown both at the national scale and in a regional study in East Anglia.

A field scale emissions inventory was developed with detailed data on agricultural practice for a 5 km by 5 km area in central England. This study provided estimates of local variability in emissions (<1 - 8,000 kg N ha^{-1} $year^{-1}$) and allowed the validation of the average assumptions built into the national scale model. Awareness of the very high variability in NH_3 emissions estimated at the field scale is critical for the assessment of impacts of NH_3 deposition from local sources, as it has been shown that the largest deposition rates occur in the vicinity of local sources and over semi-natural areas, such as forest edges. A comparison of the national 5 km grid and the field scale inventory has shown that the model results are generally robust. However a closer investigation of the underlying models and input data at both scales clearly showed that much local variability is hidden in the 5 km model results.

A major aim of this thesis was to identify sources of uncertainty in NH_3 emission inventories, both at the national and at the local scale. While previous studies only considered uncertainties in the applied NH_3 source strength estimates, all main sources of uncertainty in the model input data as well as the model assumptions were evaluated here, and quantified where possible. The main causes of uncertainty in the national inventory were found to be due to spatial aggregation effects (MAUP), and due to the spatial and temporal variability in source strength, depending on environmental conditions and agricultural practice. A quantitative assessment of the modelled spatial variability within the 5 km NH_3 emission estimates was carried out by calculating the % coefficient of variation of from the underlying 1 km results. High values of >150% were found in areas with intensive pig and poultry farming, as well as at the boundary between intensively farmed lowland areas and extensive upland and hill areas. Low values of ~20% are typical for some grassland areas with predominantly cattle farming.

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1.1. ATMOSPHERIC POLLUTION - DEFINITIONS AND PERSPECTIVE

Over the last few decades it has become increasingly apparent that air, water, soil and other natural resources cannot be exploited limitlessly without causing damage or even irreversible destruction to the natural environment. 'Sustainable development' and the precautionary principle have become key terms in environmental politics and underlie both the UK and European environmental policies (DoE, 1990; Wieringa, 1995). In this context, numerous attempts have been made to define pollution and methods of assessing and minimising its effects. Pollution can be characterised in relation to its capacity to upset naturally occurring processes and ecosystems either through toxicity or through modification of natural processes or ecosystems.

Atmospheric pollutants are a major form of diffuse environmental pollution, having effects on a whole range of spatial scales from local to global. Oke (1987) defines air pollutants as *"substances which, when present in the atmosphere under certain conditions, may become injurious to human, animal, plant or microbial life, or to property, or which may interfere with the use and enjoyment of life or property."* This definition puts the emphasis very much on the effects of atmospheric pollutants. It has to be noted that many substances, e.g. carbon dioxide, methane or some nitrogen compounds such as ammonia, are released to the atmosphere through natural processes. These are therefore only classified as pollutants when they derive from anthropogenic sources.

The life cycle of the polluting substances (Figure 1.1.) begins with their emission from a range of possible sources. Once emitted, they are dispersed through atmospheric mixing, both horizontally and vertically, by turbulent diffusion and convection. The average length of time that pollutants remain in the atmosphere varies from seconds or minutes to days for reactive pollutants (e.g. sulphur dioxide, ammonia) or months and years for inert species (e.g. carbon dioxide, CFCs). Hence pollutants may travel from a few metres to hundreds of kilometres, depending on their atmospheric residence time. Both transport distance and residence time vary depending on the pollutant reaction or removal rates. These are affected by environmental conditions, such as meteorological conditions in time and space (stability of the air, wind speed and direction), the configuration of the emitting sources (including the height above ground) and their location in relation to the surrounding area (Oke, 1987; RGAR, 1997). The primary pollutants emitted at the source may also be converted through chemical processes such as oxidation into secondary pollutants (RGAR, 1990). The main removal pathways to the earth's surface are through dry and wet deposition. The former can be defined as adsorption onto surfaces or uptake by plants upon contact with the ground, while the latter is defined as removal from the atmosphere through precipitation.

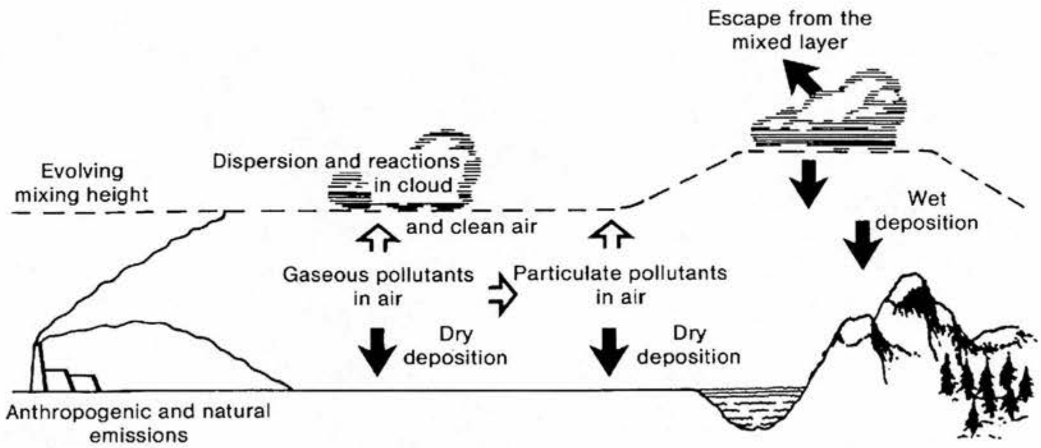


Figure 1.1. Emission, atmospheric transport and deposition of pollutants (RGAR, 1990).

The initial approach to controlling environmental pollution was through reducing overall emissions for an area by fixed percentages. It was, however, noted that the link between these 'flat rate' emission reductions and a reduction of the impacts of atmospheric pollutants was not sufficiently strong. A new approach linked the emissions to the effects through ‘critical loads’ and ‘critical levels’. Critical loads are defined as the maximum deposition that does not cause negative effects, and critical levels are defined as thresholds for direct effects of pollutant concentrations according to current knowledge (Nilsson and Grennfelt, 1988; Bull, 1991; Bull, 1995; Bull *et al.*, 1995; Metcalfe *et al.*, 1995; Bull and Sutton, 1998). The critical loads approach to environmental protection assumes that it is possible to define threshold deposition levels for pollutants, below which specific ecosystems will not experience adverse effects. Critical loads maps, developed to reflect the sensitivity of different ecosystems to particular pollutants, may be compared with deposition maps for these pollutants. This allows the identification of areas with critical loads exceedance which need greater attention, and makes it possible to devise more effective abatement strategies that reduce deposition to these areas (Hornung *et al.*, 1995). The UK government has adopted the critical loads concept as a key element of its strategy to control atmospheric deposition (DoE, 1990 and 1991).

It is apparent that the most effective form of air pollution control is to curb emissions at their source (Oke, 1987). By adopting the critical loads approach, the UK government has accepted a preventative abatement strategy designed to reduce the occasions when critical loads are exceeded and by definition a pollution event takes place. By the nature of the atmosphere as a means of distribution and dispersal, it is less simple to capture or modify the polluting substances after they have left the source and before they reach sensitive ecosystems. Therefore, abatement of atmospheric pollution must be preventative in the first instance.

Historically, atmospheric pollution has increased with the level of human interference in the environment. For instance, acid rain and the greenhouse effect were recognised as potential problems in the 19th century (Kemp, 1990). In a first wave of environmental consciousness in the 1960s (Baarschers, 1996), it was mainly the obvious detrimental effect of substances such as SO_2 or NO_x through acid rain that attracted the attention of scientists and environmental organisations and consequently the media. By the mid 1980s eventually consensus resulted in political action to abate the emissions. In 1985, the United Nations Economic Commission for Europe (UNECE) produced a legally binding document on sulphur emissions ('Sulphur Protocol'; UNECE, 1985), which required the signatories to reduce transboundary emissions of SO_2 by 30% of the 1980 levels before 1993. The 'Second Sulphur Protocol' was signed in 1994 (UNECE, 1994), requiring emission reductions for many countries of 70% by 2005 and of 80% by 2010. The 'NO_x Protocol' was signed in 1988 (UNECE, 1988). Over the last decade or so increasing attention has been directed towards ammonia (NH_3) and ammonium (NH_4^+). Currently a protocol for total nitrogen is being compiled, which not only includes oxidised nitrogen species (NO_x), but also reduced nitrogen species (NH_x) (Bull and Sutton, 1998, Grennfelt, 1998). This is because decreasing emissions from oxidised N only is not sufficient to achieve N deposition levels below the critical loads in most cases, as the contribution from reduced N is larger than that from oxidised N for much of Europe (Barrett *et al.*, 1995). Thus it is necessary to integrate efforts to produce a "multi-pollutant, multi-effect" approach (Bull and Sutton, 1998).

1.2. AMMONIA: SOURCES AND IMPLICATIONS FOR THE ENVIRONMENT

Ammonia (NH_3) is the most prevalent alkaline gas in the atmosphere. It is readily dry deposited and reactive with acidic species (such as SO_2, HNO_3 and HSO_4^-). Thus, it affects the transport distance of acidic species by forming aerosols, which have different removal rates from the precursor gases. Although NH_3 is emitted from some natural sources, the natural contribution is small compared with the losses arising from agriculture and other anthropogenic sources. Globally, the ammonia cycle is dominated by agricultural sources (e.g. Bouwman *et al.*, 1997), particularly the volatilisation from livestock manures (Figure 1.2.).

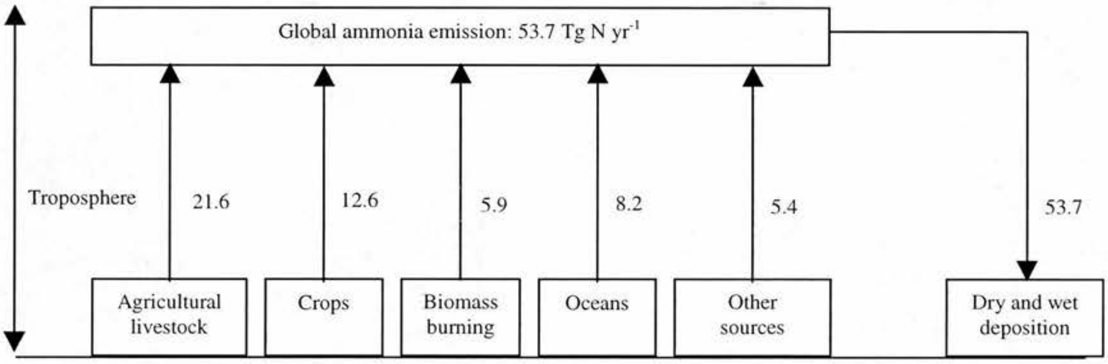


Figure 1.2. The global ammonia cycle (after Bouwman *et al.*, 1997).

The man-made sources of NH_3 within the nitrogen (N) cycle are the results of the following activities (ECETOC, 1994):

- animal husbandry, which on the one hand provides a large array of products useful to humans, but on the other hand produces large amounts of waste products (manures etc.) with their associated N emissions;
- increased N input to soils through the use of mineral fertilisers or animal manures, the growing of legumes (which can fix atmospheric N) and the movement of N from one area to another through export/import of animal feeds;

- soil usage through tillage, irrigation and drainage, which influence chemical and biological processes in the soil;

Ammonia emissions in the UK are dominated by agricultural sources (Sutton *et al.*, 1995) at about 80-85% of the total NH_3 emissions (RGAR, 1997). The spatial distribution of the main sources is significantly different from that of the sources of most other pollutants (such as NO_x or SO_2), which are mostly associated with combustion processes, industry and transport. The largest contributions to the total NH_3 emissions are made by livestock husbandry, with smaller amounts contributed by the application of mineral fertilisers to crops and grassland. The remaining NH_3 emissions, currently estimated at about 15-20%, arise from some industrial processes (e.g. fertiliser production), human perspiration, pets, sewage sludge, landfill sites, traffic, coal combustion, biomass burning, decomposition of decaying vegetation and crops, natural soils and wild animals (e.g. wild deer and sea birds; Sutton *et al.,* 1995, 1998a). All these sources are discussed in greater detail in Chapter 3. In total, agricultural sources are estimated to contribute more to deposited N (oxidised and reduced) in the UK than vehicles and power stations together (INDITE, 1994; RGAR, 1997).

As a consequence of these findings, NH_3 is now recognised as a major pollutant. Its contribution to a range of environmental problems may be summarised as follows:

- In very high concentrations NH_3 is toxic to plants, humans and animals. In the vicinity of strong sources, such as intensive livestock farming units, it can directly damage the vegetation when critical levels are exceeded (e.g. Van der Eerden *et al.*, 1994).

- Increased deposition of NH_3 leads to N eutrophication of oligotrophic ecosystems, such as unmanaged forests, moorlands and heathlands. This may result in changes in the composition of the vegetation with a subsequent loss of species diversity, leading to the local extinction of nitrogen sensitive plant species. These eutrophicating effects have been described for the Netherlands (Roelofs, 1986; Van der Eerden *et al.*, 1998), where shifts from heath and peatlands to grassland systems have been well documented, as well as in the UK (Hornung *et al.*, 1995; Pitcairn *et al.*, 1995, 1998).

- Ammonia plays a significant role in the processes involved in acid deposition, especially where large quantities of SO_2 and NO_x are present (RGAR, 1990). Consequences of this are soil acidification (e.g. Sutton *et al.*, 1992) as well as acidification of water bodies (e.g. Harriman *et al.*, 1995; Hornung *et al.*, 1995). Soil acidification subsequently increases the fluxes of climate relevant gases such as N_2O to the atmosphere (Butterbach-Bahl *et al.*, 1998).

- Further reported effects of N deposition are increased sensitivity to natural stress factors, such as frost (Aronsson, 1980), attacks by fungi etc. (Roelofs *et al.*, 1985), and nutrient imbalances, such as losses of foliar Ca, Mg and Zn (Boxman and Van Dijk, 1988; Boxman *et al.*, 1991). Nitrogen deposition also causes decreases in the fine root mass and deterioration of the waxy cuticle (Van der Eerden *et al.*, 1992; Sutton *et al.*, 1993b).

Some of these effects do not become apparent within a short period of exposure events, but only become visible after an accumulation of deposited N has taken place. Most of the changes occurring are therefore gradual. Hence it is important to identify problem areas before too much damage is done, as remedial actions are difficult (Van Egmond, 1998; Grennfelt, 1998; Cowling *et al.*, 1998).

One of the main characteristics of NH_3 pollution is that the deposition of volatilised NH_3 occurs to a large extent close to the sources (RGAR, 1990; Sutton *et al.*, 1998b), i.e. a considerable proportion of the NH_3 emitted from a source does not travel further than about 5 km. Singles (1996) estimated that approx. 5-50% of all NH_3 emitted is deposited within a 5 km radius of the source. The large spatial variation in NH_3 emissions is mirrored by an equally variable spatial pattern regarding deposition and effects. This becomes even more apparent when the closeness of strong NH_3 sources and highly sensitive ecosystems is highlighted through the comparison of deposition and critical loads maps. In order to be able to quantify and predict impacts and develop abatement measures effectively, it is therefore extremely important to take the relative spatial location of sources and sinks into account.

1.3. EMISSION INVENTORIES

1.3.1. General requirements in developing inventories

Planning and implementing sensible air pollution abatement measures for the best possible protection of the environment requires long-term observation of the environmental situation as well as the clear presentation of real or hypothetical scenarios, which are the result of modelling of alternative strategies. Such a modelling approach helps to recognise potential effects of planned abatement measures before implementation.

Emission inventories are an important tool for scientists, environmental planners and decision-makers. In general, emission inventories can be defined as tools for taking stock of the emission sources of one or more pollutants in a quantitative manner. In most cases, the aim is to take account of all the sources within an administrative area. The extent of the area covered determines whether the inventory is at the local/regional level, national level or at the global level. The local/regional level can vary from a single farm to a town, city or region. The extent of the area covered has practical implications for the scale/resolution at which an inventory is compiled.

Emission inventories as quantitative databases on emissions from an area can serve many purposes:

- to provide pollution information to the environmental agencies and individuals involved (including the public)
- to identify the activities responsible for pollution and to highlight the most important emission sources. This is of great significance when the (ultimate) objective is to set priorities for abatement measures.
- to aid the setting of explicit objectives and constraints
- as a tool for local/regional or national planning, allowing scenario testing of proposed new developments. It may also prove useful in combination with inventories from neighbouring regions or countries to compare and improve the emission situation, and to set up transboundary emission control programmes.

- as a tool for the assessment of the potential environmental impacts and implications of different abatement strategies, including the evaluation of environmental costs and benefits of different policies

- to aid with the monitoring of the state of the environment after the implementation of abatement policies, as a check for the achievement of targets, and to ensure that those responsible for the implementation of abatement policies are fulfilling their obligations.

- Spatial emission inventories provide the basis for atmospheric transport and deposition models, which allow the effects of the emitted pollutants to be studied.

The development of an emission inventory will normally contain the main phases/steps outlined in Figure 1.3.

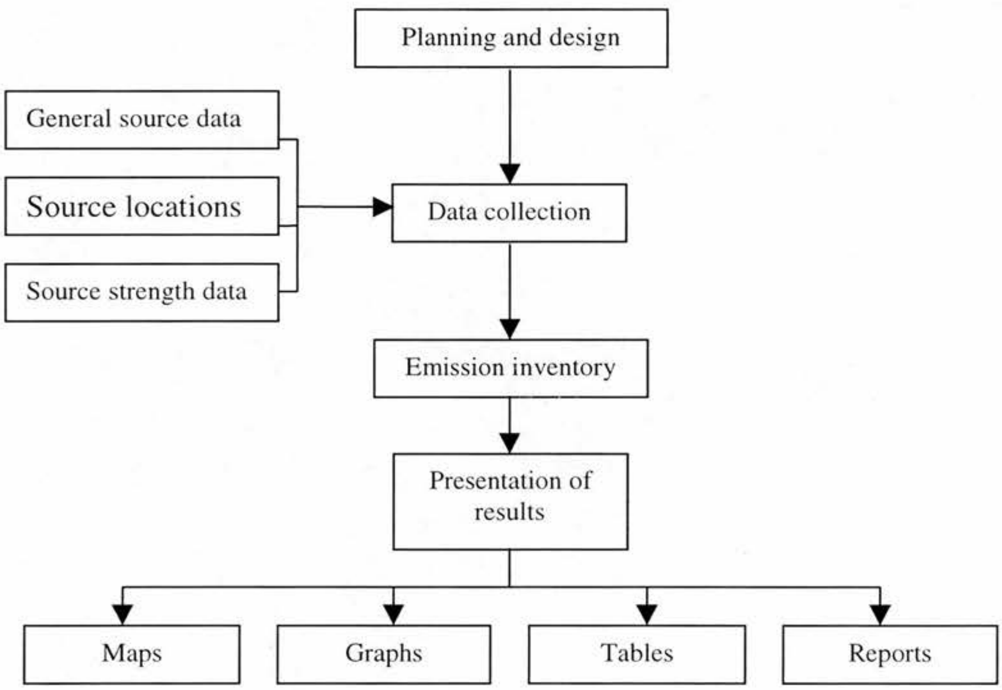


Figure 1.3. Phases of the development of an emissions inventory.

The aims and objectives of an emission inventory vary depending on the scale and spatial extent of the study, the pollutant(s) covered, the data and resources available etc. The relevant data for the emission sources to be included in the inventory (such as location, emission source strength, capacity, operating conditions and times, source-specific measurements where available including measurement methods etc.) have to be collected. This may be accomplished through dedicated surveys, the compilation of already available statistical data or a combination of both. Such data may include information from the owners, producers or operators of the emission sources as well as regionally or nationally collected data such as censuses or sample surveys. Subsequently, the data have to be manipulated and stored in a suitable system with efficient retrieval and modelling capabilities.

1.3.2. Approaches for calculating emissions

Emission inventories may be developed using several different approaches (TFEI, 1996):

- from measurements at the sources
- through the use of average emission factors for each source type
- through more detailed process-based modelling, with component emission estimates

In most cases, it is not possible to measure emissions directly at all the individual sources that need to be included into the inventory. However, this approach is sometimes used for partial emission inventories or inventories of certain source sectors, such as large industrial sites. In the UK, for instance, the Environment Agency maintains the Chemical Release Inventory (CRI), a register for all major pollutants in England and Wales. The emissions of the relevant pollutants for each site are in many cases measured directly at each plant.

In practice, the emissions are estimated in most cases with the help of average 'emission factors' or 'emission source strength data' (TFEI, 1996). These are based on measurements made at representative sources or sites. In the simplest case, the total emission (E) is calculated as the product of the number of emission sources (n) and the emission source strength per average source (f) for a certain specified time/duration.

$$E = n * f$$

For example, the annual emissions of SO_2 from a coal-fired power plant (in grams, kg or tonnes per year) can be calculated from

- the annual coal consumption (in tonnes coal per year) and an average emission factor (in grams SO_2 emitted per tonne of coal consumed) or

- measured SO_2 emissions (e.g. in grams per operating hour) and the number of operating hours per year.

Similarly, the annual emissions of NH_3 from a dairy farm per year may be calculated by extrapolating daily measured average emissions to annual emissions or by multiplying the number of cows with average annual emission estimates per cow. In practice, the calculations may be more complicated, but they follow the basic principles outlined above. For instance, Cowell (1998) developed a process-based model for NH_3, which uses component emission factors throughout to allow for different livestock management systems.

Mean source strength data estimate the amount of emission to be expected on average for a certain type of source. It should be noted that, in many cases, the real emissions from any one source will differ, sometimes substantially, from the mean estimated ones, depending on the particular circumstances. Source strength may be influenced by specific characteristics of the source itself (e.g. quality of fuel used in a combustion process or nitrogen content in livestock feeds) or by external circumstances such as weather/climate (temperature, humidity etc.). Therefore, some estimate of uncertainty should be taken into account during the evaluation of the results. Best- and worst case scenarios may prove helpful in this instance to indicate this uncertainty and help our understanding of the nature of the processes and the potential errors in the calculations.

1.3.3. Development of spatial inventories

The problems of atmospheric pollution cannot be fully addressed without taking the spatial component into account. Thus the tabulated total emissions are ideally accompanied by cartographic representations of the inventory. A spatially distributed emissions inventory will provide an instant overview over the areas most in need of improvement, rather than aggregated sums for the whole inventory area. For instance, mapped ammonia emissions are necessary to identify source and sink areas, and to get an understanding of their proximity in space. Spatial representation is also essential if any spatially distributed atmospheric transport, deposition or critical loads models are to be attached to the emission inventory. In most cases, this representation will be based on a raster or grid data set. In order to be able to provide the links, it is necessary to integrate a spatial view of the sources from the beginning of the development of the inventory. This requires geo-referencing of the emission sources to be introduced from the start.

For this purpose, all emission sources may be disaggregated into three main categories: point, line and area sources. It is important to distinguish between the actual emission sources themselves and their representation within models. For instance, a model may define a large city as a point source or an area source, depending on the scale and resolution of the study. In a gridded model, all sources are by default treated as area sources.

A point source in a model is a single identifiable emission source that is large enough to merit separate attention. Depending on the scale and resolution of the inventory, it may consist of a single power station stack (local scale) or, on an international/global scale, a large city.

Linear sources are mostly associated with roads, railways, air corridors or similar features. In general, they would be associated with sources large enough to merit the effort of collecting or assigning specific data with them, such as motorways or other main roads.

An area source, by definition, is an emission source which extends beyond a single point, as defined by the resolution of the study. This may be > 1 metre, > 1 km, or 5 km etc. The contributors to any single area source cannot be differentiated to a finer resolution, either due to lack of data or due to the cost of doing so outweighing any benefit. An area source contains many similar sources, which emit roughly the same amounts of the pollutant in question, i.e. the emission within each area source is assumed to be more or less homogeneous. Area sources may comprise anything from a grazed field (local scale) to large industrial areas, housing estates or agricultural areas, depending on the scale and resolution of the study. In many cases, the individual sources are much smaller or more diffuse than point or line sources and would not merit individual measurements or georeferencing. Examples for this are emissions from traffic on smaller roads in suburbs that are not given the status of linear sources or emissions from domestic combustion that are not important or different enough to be stated as individual point sources.

1.3.4. Development of temporal resolution in emission inventories

Emission sources also have a strong temporal component. While industrial and traffic emissions such as SO_2 or NO_x tend to follow daily and weekly rhythms (rush hour, weekend traffic, operating hours of large industrial plants etc.), agricultural emissions are also prone to seasonal fluctuations (Asman, 1992b; Kruse *et al.*, 1989). For instance, the spreading of manures from livestock housing onto suitable fields, livestock grazing or fertiliser applications predominantly take place during specific times of the year. Furthermore, the emission source strength of ammonia varies over time, dependent on environmental factors such as temperature, precipitation, humidity etc (see Section 2.2.). These variations are not considered in many emission inventories, with most studies providing average annual emissions.

1.3.5. Historical development and examples of emission inventories

Historically, the first emission inventories were created in the 1960s (Ahamer, 1989) in the United States (Chicago 1966; New York, San Francisco and 88 further US cities until 1968). In Europe, the first SO_2 inventory was produced for Berlin in 1965. In the following paragraphs, a few examples of existing emission inventory projects are outlined and their approaches discussed.

The APIS (Air Pollution Information System) project developed a prototype for the processing of atmospheric pollutant emission data and monitoring in the surroundings of power stations and large industrial areas (Rinaldi *et al.*, 1993; Trozzi and Vaccaro, 1993; Spalla and Viola, 1993). The aim of APIS is to provide an integrated model which helps to control the air quality in the area. A pilot project was developed for a power plant near Piombino, Italy, and an outline of this is shown in Figure 1.4.

The emission inventory is derived from statistical data and presented in tabular format as well as mapped onto grid squares. There is a strong spatial component in the project, integrating thematic maps, aerial photography, satellite imagery and the data from the monitoring system. The monitoring network measures air quality as well as meteorological factors (temperature, wind speed and wind direction) in short time intervals. With the results of the emission inventory, diffusion/dispersion models are run to calculate air concentrations.

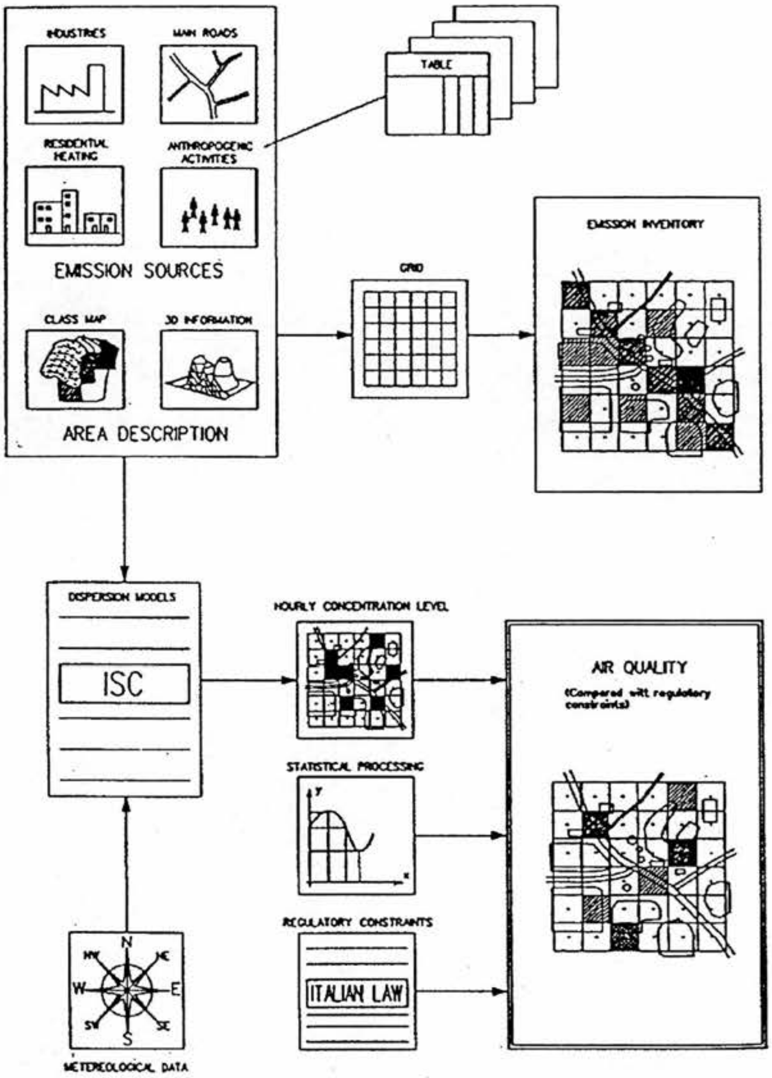


Figure 1.4. The APIS Project. Source: Rinaldi *et al.* (1993).

In Estonia, several projects are under way to repair some of the existing environmental damage (Perrett and Saare, 1990). Initially, a pilot area in north-eastern Estonia was selected, for which base data on built-up areas, roads, railways, waterbeds, moorland and other environmentally sensitive areas were collected. Building on these data sets, several projects were designed. One of them involved the preparation of an emissions inventory with the purpose of introducing high environmental taxes for any source exceeding the maximum allowed emissions. Taxes depend not only on the amount emitted or the toxicity of the pollutants, but also on the size and landcover types of the area affected. This requires a strong spatial component in the emission inventory. In association with this inventory, another project involved the collection of precipitation samples, from which acid deposition maps were generated. By spatially overlaying the results with the base data sets, the causes, effects and the spatial connections between sources and sink areas were studied and used to help support decisions about abatement measures. A weakness of this study was that no atmospheric transport models were included to provide the link between emissions and effects.

The Dutch National Institute of Public Health and Environmental Protection (RIVM) has not only been instrumental in the generation of a sophisticated emission inventory for the Netherlands, but also in coordinating European efforts. A main aim of the work at RIVM is the development of an environmental database for Europe (Van Beurden and Scholten, 1990). One of the projects involved a risk assessment of the effects of atmospheric pollution on public health in Europe (Van der Veen, 1992). The aim of this project was to enable a more precise, detailed and regionally differentiated assessment of the available data on atmospheric pollution. This required the integration of large amounts of already collected statistical data from all European countries, with the added complications of widely differing data quality, dates (year) of collection and level of aggregation of the data (Van Beurden and Scholten, 1990). A major result of this project was a model of deposition of atmospheric pollutants (Figure 1.5.), derived from the emissions inventory, at national, European and global scales (Kusse *et al.*, 1993).

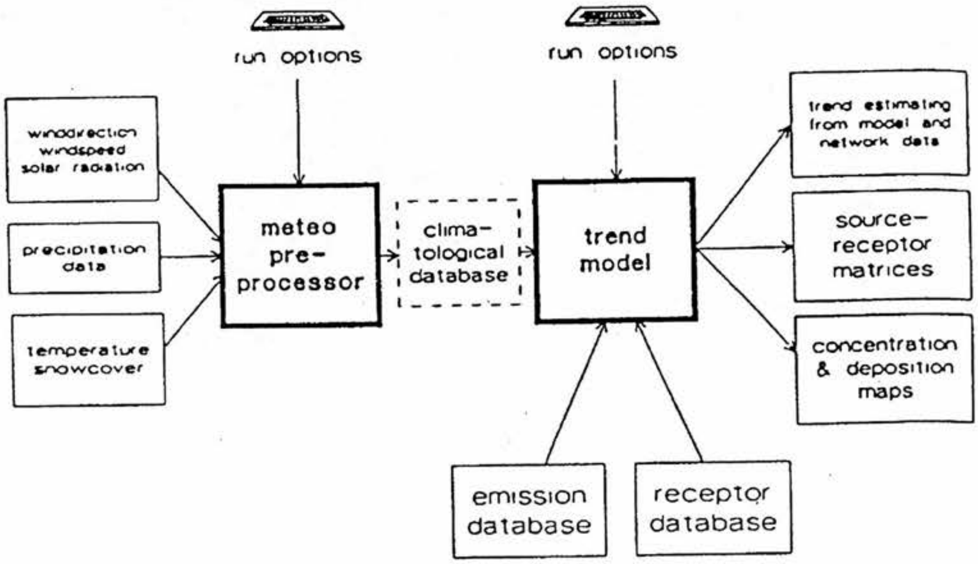


Figure 1.5. The RIVM model. Source: Kusse *et al.*(1993).

The parameters for the spatial distribution and atmospheric transport of the emissions are defined within the system, while the pollutant specific parameters are defined externally. This allows the model to be used flexibly for different pollutants. Because of the large overhead in computing resources, the meteorological parameters are calculated in advance for all standard weather situations. The results of this are air quality maps and risk maps. The latter are combined with maps of population density to set priorities for abatement measures.

Recently a joint CORINAIR/EMEP 'Atmospheric Emission Inventory Guidebook' (TFEI, 1996) has been published by the *Task Force on Emission Inventories* (TFEI). The aim of this guideline is to assist member countries of the UNECE in developing their own national emission inventories for all relevant pollutants by providing them with methods and source strength information (Bull and Sutton, 1998; Van der Hoek, 1998). The individual national inventories are then submitted to EMEP, the *Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe* (e.g. Berge *et al.*, 1995), for modelling and mapping of emissions, atmospheric transport and deposition at the European scale. Figure 1.6. shows a map of European ammonia emissions from agricultural livestock for 1989.

The projects briefly outlined above are just a few examples of past or ongoing activities at different scales within Europe. All the examples stress the following two points: the importance of the spatial view and the inseparable links between emissions, transport and effects of atmospheric pollutants. Only when the spatial distribution of the effects is taken into account, are any sensible and effective abatement measures possible. The deposition of pollutants and their effects in sink areas are intrinsically linked with the source areas through atmospheric transport mechanisms. Hence the development of spatially resolved inventories is a primary requirement for quantifying air pollution impacts and developing abatement strategies.

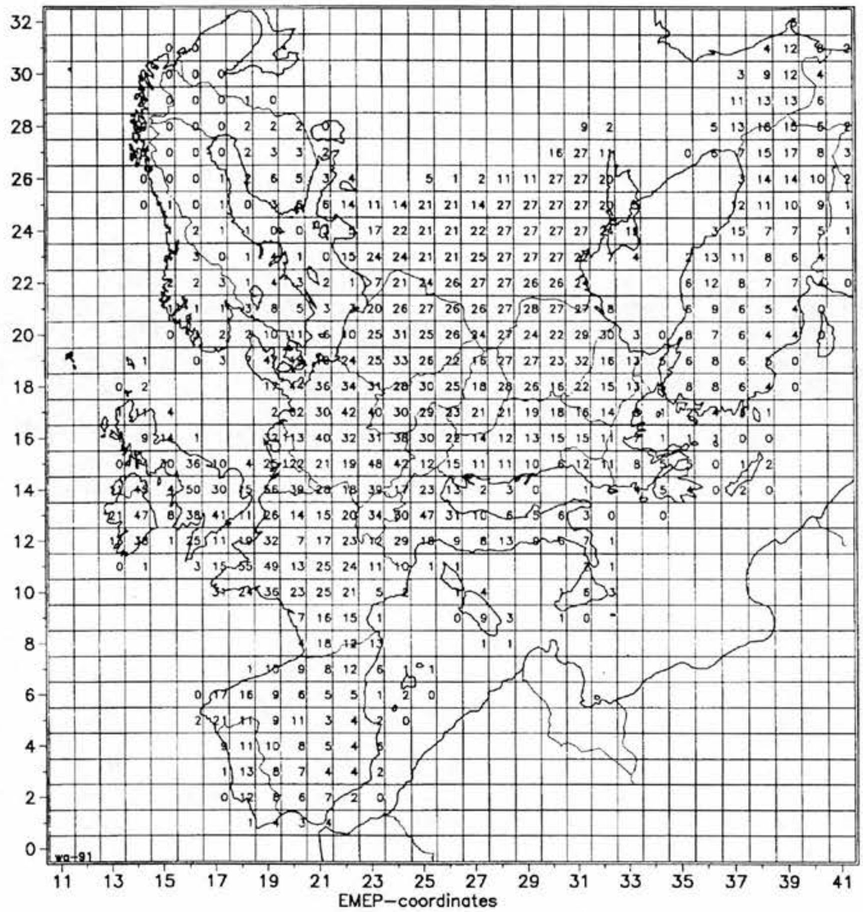


Figure 1.6. EMEP: Ammonia emissions from agricultural livestock in Europe for 1989 (in 1000 t NH_3 $year^{-1}$). Source: Asman (1992b).

1.4. AMMONIA EMISSION INVENTORIES

A large number of emission inventories has been produced specifically for NH_3 since the 1980s, for single European countries (Buijsman *et al.*, 1984 (Netherlands); Kruse *et al.*, 1989 (UK); Erisman, 1989 (Netherlands); Möller and Schieferdecker, 1989 (GDR); Asman, 1992a (Denmark); Fekete, 1992 (Hungary); Pipatti, 1992 (Finland); further inventories: see Klaassen, 1992a) as well as for Europe as a whole (Buijsman *et al.*, 1987; Asman, 1992b; Klaassen, 1992b; ECETOC, 1994), North America (Cass *et al.*, 1982 (USA); Geadah, 1985 (Canada); Heisler *et al.*, 1988 (USA)) and Australia (Denmead, 1990). Most of these studies are based on estimated livestock numbers as the main source of NH_3 emissions, combined with estimates of ammonia emission source strength data per animal for the main livestock categories. In addition, smaller amounts are included to account for emissions from the application of fertiliser to crops and grassland and other minor sources (Sutton *et al.*, 1995).

The emissions from livestock farming arise mainly from the decomposition of urea and uric acid in manures (see Section 2.2.). Emissions occur from livestock housing, manure storage (farmyard manure and slurries), land spreading of manures and from grazed pastures. The source areas, farmyards (with animal housing and manure storage facilities), fields and pastures are spread across the landscape in historically developed patterns and result in highly variable patterns of NH_3 emissions in space, on a local level as well as on a national or international level.

At a national level, there are areas that are particularly suited to certain types of agriculture, which can be as diverse as intensive dairy farming, pig production or extensive hill sheep farming. These specialised areas are represented by distinctive spatial patterns of NH_3 emissions, showing, for instance, relatively small and evenly distributed sources for extensive hill farming areas and the large spikes typical for intensive poultry and pig farming areas. The high density of animals in intensive livestock farming (especially poultry and pig farms) confined to very limited areas results in large point sources of NH_3 in the landscape, which emit at high levels throughout the year. At the other end of the scale, grazed pastures are relatively weak and diffuse NH_3 sources, especially when they do not receive large amounts of N fertiliser. Hence, if NH_3 emission inventories are only presented in a tabular format, the aspects and consequently effects of this spatial variability cannot be taken into consideration for the design of effective abatement strategies.

Some of the above mentioned inventories also provided spatially distributed NH_3 emission estimates, in raster/grid format. Examples for this are the European NH_3 emission maps by Buijsman *et al.* (1987) and Asman (1992b) at resolutions of 150 km by 150 km and 75 km by 75 km (Figure 1.6. above). At a finer spatial resolution of 5 km to 25 km, many mapped national inventories have been generated in addition to the non-spatial inventories (Buijsman *et al.*, 1984; Erisman, 1989 (Netherlands); Kruse *et al.*, 1989 (England); Asman, 1992a (Denmark); Fekete, 1992 (Hungary); Pipatti, 1992 (Finland); Graf, 1992 (Eastern Germany); Sutton *et al.*, 1995 (Great Britain)). These more detailed NH_3 emission maps are particularly important because of the extreme spatial variability of both emissions and deposition of NH_3 (Sutton *et al.*, 1993a). The improved spatial resolution of the emission sources and the reliability of the data on source locations are necessary for identifying and spatially pin-pointing the inherent spatial variability in NH_3 impacts. It is also essential for designing and implementing of effective abatement measures. Any estimated deposition or critical loads exceedance maps derived from atmospheric transport modelling can only be as good as the estimated emission maps provided.

It is, however, not only the spatial location of NH_3 emission sources that is critical in spatially distributed inventories: estimating NH_3 emissions for an area, whether spatially distributed or not, depends heavily on reliable source strength data. Whereas the estimated SO_2 and NO_x emissions in the UK are believed to have a ± 20% margin of uncertainty (RGAR, 1990), estimates of ammonia emissions differ much more widely. Total NH_3 emissions for the UK have been estimated by recent studies at between 201 kt N year^{-1} (BBSRC, 1997a; from agricultural sources only) and 444 kt N year^{-1} (ECETOC, 1994; including non-agricultural sources). These studies (Table 1.1.) have large uncertainty margins associated with their estimates (INDITE, 1994; Sutton *et al.*, 1995). Sutton *et al.* (1995) examine the key differences between different studies in detail, with regard to total emissions as well as to the contributions from the different sources, and suggest ranges of uncertainty and current best estimates. In 1995, official NH_3 emissions for the UK (DoE, 1995) for all major sources were agreed by a panel of experts for submission to EMEP. These have recently been updated (BBSRC, 1997b; Sutton and Fowler, 1998). Despite this, large uncertainties still remain regarding emission source strength estimates. These are discussed in more detail in Chapter 3.

In the UK, the first spatially distributed NH_3 emission inventory for agricultural sources was produced for England and Wales (Kruse, 1986; ApSimon *et al.* 1987; Kruse *et al.*, 1989), at a grid resolution of 10 km by 10 km (Figure 1.7.). More recently, Eager (1992), Sutton *et al.* (1995) and Dragosits *et al.* (1996b) produced inventories for Great Britain at a resolution of 5 km by 5 km. These are compared with the new inventory developed in this study in Chapter 6.

In Chapter 7, temporal change of UK NH_3 emissions is modelled and discussed for the past (1969, 1988 and 1996), as well as projected for the future, by developing abatement scenarios. The new national inventory is then compared and validated with a field scale inventory for a 5 km by 5 km study area, regarding both spatial variability of NH_3 emissions within a 5 km gridsquare and spatial variability of emission source strength (Chapter 8). Furthermore, uncertainties in the spatial emission inventory are identified and quantified where possible (Chapter 9). This includes uncertainties in the model input data, the emission source strength estimates, and in the model assumptions, as well as temporal uncertainties.

Table 1.1. Estimates of NH_3 emissions in the UK (Sutton *et al.*, 1995; RGAR, 1997)

Author	Reference year	Emission (kt NH_3-N yr^{-1})	Emission sources
Healy *et al.* (1970)	mid 1960s	70-105	Agriculture; other sources included in higher estimate
Hood (1982)	1978	595	Agriculture
Fisher (1984)	1977	415	Agriculture
Buijsman *et al.* (1987)	~ 1980	334	Agriculture
Ryden *et al.* (1987)	mid 1980s	355	Grassland and livestock emissions only
Kruse *et al.* (1989)	1981	371	Agriculture (GB only)
Jarvis and Pain (1990)	~ 1988	186	Agriculture
Asman (1992b)	1989	385	Agriculture
Klaassen (1992b)	1987	405	Agriculture
Eggleston (1992)	1980-88	434-461	Agriculture and other sources; range of best estimates for different years 1980-1988
ECETOC (1994)	1990	489	Agriculture and other sources
Sutton *et al.* (1995)	1988	371 (190-599)	Agriculture and other sources (uncertainty estimates in brackets)
DoE (1995)	1993	260	Agriculture and other sources
BBSRC (1997a)	1993	201	Agriculture
BBSRC (1997b)	1996	226	Agriculture

Figure 1.7. Ammonia emissions from agricultural livestock and fertiliser application in 1981 for England and Wales. Key (kt N as NH_3 emitted per 100 km^2): white, 0-25% (0-88 t N); light grey, 25-50% (88-160 t N); dark grey, 50-75% (160-255 t N); black, 75-100% (255-1451 t N). Source: Kruse *et al.* (1989).

1.5. THESIS PLAN

The aim of this thesis is to review the present situation regarding spatially distributed NH_3 emission inventories and source strength estimates in the UK, and to develop a new and improved spatial emissions model, thus providing more realistic key input for atmospheric transport and deposition models. This includes updating the reference year for the national 5 km grid inventory to 1996 and providing a spatially distributed inventory of Northern Ireland for the first time. Furthermore, the local variability of NH_3 emissions at a large scale is studied, and uncertainties encountered when developing spatial NH_3 emission inventories are investigated in detail.

- This chapter has introduced NH_3 as a major issue in the context of atmospheric pollution and current efforts towards abatement. The main sources and impacts of NH_3 have been outlined briefly, and the need for spatial emission inventories discussed.
- Chapter 2 considers aspects of UK agriculture relevant to NH_3 emissions. This includes a general introduction to the basic principles of NH_3 volatilisation, and describes how the magnitude of emissions is influenced by environmental factors (climate, topography) and agricultural practice.
- Chapter 3 provides an overview of the current state of research into ammonia emission source strength ("emission factors"), by comparing and critically reviewing estimates from recent inventories for the UK and Europe. The focus is on agricultural livestock as the main source of ammonia emissions, but also includes emissions from fertilisers and other miscellaneous sources.
- Chapter 4 addresses some general aspects of the proposed new approach to modelling the spatial distribution of NH_3 emissions for the UK. This includes the choice of model input data and a suitable implementation environment.
- Chapter 5 describes the new UK model and compares and contrasts the new methodology with that employed in previous studies.
- Chapter 6 presents and analyses the UK model results for agricultural and other miscellaneous sources in detail, especially regarding contributions from different source sectors and their characteristic spatial patterns.

- In Chapter 7, temporal change of UK NH_3 emissions is modelled and discussed for the past (1969, 1988 and 1996), as well as projected for the future by developing abatement scenarios.
- In Chapter 8, a local NH_3 emissions inventory is developed. The results are used for an investigation of the large scale local variability of NH_3 emissions, regarding aspects of spatial variability as well as the variability of emission source strength within the study area. A comparison of this detailed local inventory with the average conditions applied in the national inventory is also undertaken, to assess the validity of the model assumptions regarding average practice.
- In Chapter 9, the main sources of uncertainty encountered in modelling and analysing NH_3 emission inventories are discussed. This includes uncertainties in the spatial data sources, the emission source strength estimates and the model assumptions as well as temporal uncertainties.
- Chapter 10 provides several case studies that show how the basic model described in Chapters 4-7 may be refined and improved in future work, by, for example, developing spatially variable emission source strength estimates instead employing UK average values.
- Finally, Chapter 11 summarises the work undertaken in the course of this thesis, and reflects on the wider implications of the results, by pulling the different aspects considered in Chapters 1-10 together and presenting the main conclusions. This also includes validating the UK model results against an independent set of measurements and reviewing deposition and critical loads maps derived from the new inventory.

1.6. SUMMARY

Atmospheric pollution is a significant problem, causing many detrimental effects and affecting humans as well as the environment. Over the last two decades, scientists and decision makers have become increasingly aware of pollution through reduced nitrogen species, mainly ammonia (NH_3) and the reaction product ammonium (NH_4^+). Ammonia is the predominant alkaline gas in the atmosphere and originates to a large extent from agricultural sources, in particular from livestock manures and mineral fertiliser applications. Among other effects, it causes the eutrophication of nitrogen sensitive ecosystems and plays a major part in the acidification of soils and waterbodies, in combination with sulphur species and oxidised nitrogen species.

Effective abatement measures can only be devised if the cycle of emission - atmospheric transport - deposition and effects is seen as a whole. It is therefore essential to have reliable quantitative information about emissions. It is also necessary to identify the relative spatial location of sources and sinks, in order to be able to set target-oriented abatement measures. This spatial perspective is extremely important with regard to ammonia, as it tends to deposit close to its sources and therefore shows large spatial variations over the country.

Emission inventories may be defined as tools for quantitative stocktaking of all the emission sources in a specified area. In general, the emissions are calculated as the product of average emission source strength data (for each type of pollutant) and the number of sources. If the inventory provides a spatial component in addition to tabulated results, it gives a very quick and effective overview over potential problem areas, which is necessary for setting priorities for abatement efficiently. Spatially distributed emission inventories are essential input data for atmospheric transport and deposition models.

It is the aim of this study to provide a new NH_3 emission inventory for the UK with an improved spatial distribution of NH_3 sources, and thus to provide an updated as well as more realistic and complete basis for transport and deposition modelling, and consequently effects-based abatement strategies. Furthermore, it is a primary objective to investigate the uncertainties inherent in spatial NH_3 emission inventories and devise ways to quantify and understand these uncertainties, and thus provide a starting point for further improvements.

Chapter 2
Perspectives of UK agriculture affecting ammonia emissions

2.1. INTRODUCTION

Agricultural activities are governed to a large extent by relief, soil and climate. In addition, the size, shape and layout of farms and fields, ownership conditions, the availability of equipment, capital and labour, the location of markets and the attitudes of farmers all have an influence on the way agricultural land is used (Coppock, 1976a). Many of these features have their roots in the historic evolution of modern agriculture. This trend has been accentuated since the entry of the UK into the European Community (1973), with the subsequent intervention of the Common Agricultural Policy (CAP).

The agricultural landscape of the United Kingdom is very diverse with fertile lowland areas, large upland tracts and vast areas with predominantly extensive hill farming. The drier eastern side of Britain is better suited for arable cropping, whereas the more humid western parts are excellent for grassland and livestock production. The variations in the natural and human factors influencing agriculture occur at the local as well as at the regional scale. This diversity results in a multi-facetted mosaic of agricultural activities, not only at a local (field) level, but also at a regional or national level. This diversity is mirrored in the source distribution, source strength, timing etc. of NH_3 emissions. For instance, no two dairy cattle farms have the same emissions per animal per year. Differences could be due to animal breed, age, feed composition (especially the N content), grazing conditions, housing type and conditions, waste storage, weather conditions, waste spreading practice and machinery etc.

Any model trying to provide a realistic picture of the spatial distribution of NH_3 emissions needs to incorporate knowledge about current agricultural practice in general as well as geographical or historic differences in these practices. As mentioned earlier (Chapter 1), one of the main problems of emission inventories is that they generally apply average conditions. In the case of NH_3, some of the factors influencing these averages, such as temperature or fertiliser application rates, have distinctive and definable spatial distributions. In principle these could be incorporated into the emission model to improve it further, thus not only providing more realistic output, but also the prospect for quantitative assessment of the spatial variation of certain emission source strength estimates (see Chapters 9 and 10).

In this chapter, the physical basis of NH_3 emissions from agricultural sources is first considered, followed by a discussion of how this interacts with environmental factors influencing agriculture in the UK and agricultural practice. The aim is to provide a general overview of UK agriculture with respect to NH_3 emissions, taking into account average conditions as well as more specific situations. Special care is taken to point out variations in agricultural practice, as far as they may be relevant to modelling the spatial distribution of NH_3 emissions.

2.2. BASIC PRINCIPLES OF AMMONIA VOLATILISATION FROM AGRICULTURAL SOURCES

Consideration of the basic biochemical processes and reactions of NH_3 is helpful for understanding the underlying principles determining NH_3 source strength. Examples are why annual emissions from livestock grazing for part of the year may increase when the housing period is extended, or why emission rates are expected to be higher in warmer parts of the country. The basic principles guiding NH_3 emissions are shown in the combined water solubility and dissociation equilibria (Equation 2.1.; Sutton *et al.*, 1993b). Ammonia gas (NH_3) exists in equilibrium with ammonium (NH_4^+) in aqueous solution, with the partitioning being pH dependent. Thus, the more water is available for a given amount of NH_4^+, the higher is the possibility for the NH_3 to dissolve, and hence the smaller the emissions.

$$[NH_3(gas)] = 10^{(1.6035-4207.62/T)} \frac{[NH_4^+]}{[H^+]} \qquad [2.1.]$$

The following main points can be explained with this equilibrium:

(1) In drier conditions, the concentration of NH_4^+ increases in the solution due to less water being available, and higher emissions are the result. This explains, for instance, why emissions are higher from fertiliser applications in dry conditions.

(2) Temperature (*T*) and pH value of the solution influence the equilibrium in the following manner: with rising temperatures the concentrations of aqueous NH_3 double roughly with every 5°C increase in temperature, due to reduced solubility. This is shown in Figure 2.1. The main effect regarding ammonia source strength, which can be explained through the influence of temperature, is that increased temperatures cause an increase in emissions.

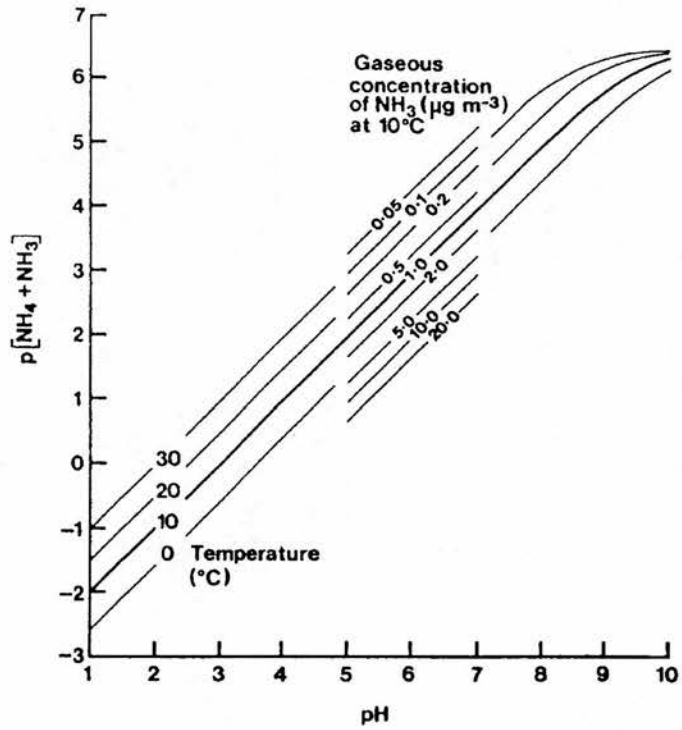


Figure 2.1. Relationship of equilibrium total NH_3 and NH_4^+ in water for different pH, temperature and air concentrations of NH_3 . Aqueous ammonia concentrations are plotted as $p[NH_3 + NH_4^+] = -\log_{10} [NH_3 + NH_4^+]$ for comparison with pH. (from: Sutton *et al.*, 1993b)

(3) $[NH_4^+] / [H^+]$ is the ratio of NH_4^+ over acidity, where $pH = -\log_{10} [H^+]$. Thus with higher acidity (i.e. smaller pH) the NH_3 emission rate decreases. This explains why acidifying additives can be used as a means to reduce NH_3 emissions from stored slurry (e.g. Husted *et al.*, 1991).

As a secondary effect, cooler temperatures lead to more livestock housing, which in turn leads to higher emissions. For instance, emissions from animals that are mainly outdoors on pastures, such as sheep, are very low. Only about 6-8% of the total N they excrete is volatilised as NH_3 (see Chapter 3). Emissions from part-housed and part-grazed animals, such as dairy cattle, are higher, with about 20-30% of their total N excretion volatilised. Volatilisation rates from purely housed livestock such as fattening pigs and poultry are highest, with losses of over 30% of the excreted N.

The reasons for these higher emissions with higher housing rates are again linked to the saturation of liquid surfaces (see above). The greater the surface capacity, the smaller the volatilisation rates of NH_3, and with less surface capacity emission increases.

Grazing systems can absorb and hold a larger fraction of the NH_3 since concentrations are low and well dispersed in time and space. In contrast, housing systems as well as stored manure and landspreading provide conditions with very high concentrations of NH_4^+ in solution, with the result that emission rates are much larger. Thus spreading N sources thinly onto an absorbing surface such as a grass sward results in smaller emission rates, due to maximising of the capture capacity and minimising of individual doses on an area basis. This contrasts with concrete surfaces in animal houses, where there is limited space for absorption, or manure spreading, where the large applications occurs within a very short time span.

Ammonia emission processes vary for different forms of agricultural N, such as urea, uric acid and mineral fertilisers such as ammonium nitrate. The enzymic hydrolysis of urea or ureolysis (see e.g. Jarvis and Pain, 1990), i.e. the reaction of urea and water, results in urea being broken down into ammonium, bicarbonate and hydroxide ions (Equation 2.2.).

$$CO(NH_2)_2 + 3H_2O \xrightarrow{urease} 2NH_4^+ + HCO_3^- + OH^- \qquad [2.2.]$$

According to Equation 2.1. above, large increases in NH_4^+ in the solution reduce the acidity, and large NH_3 emissions follow. This is the basis for significant NH_3 emissions from livestock manures, in particular urine, which contains a large proportion of its N in form of urea. Urea in mineral fertilisers (see Sections 2.5. and

3.3.) reacts in the same way. Dissolution of ammonium nitrate, on the other hand, does not give rise to large pH increases, and therefore ammonia emissions are smaller than from urea.

A reaction similar to ureolysis occurs with uric acid, which is a main constituent of poultry manures (Equation 2.3.; after Hutchinson, 1950). Uric acid reacts with water and oxygen to produce urea and glycoxylate. The urea is then further hydrolysed to NH_3 according to Equation 2.2.

$$C_5H_4N_4O_3 + O_2 + 2(H_2O) \rightarrow 2(CO(NH_2)_2) + COHCO_2 \qquad [2.3.]$$

A corollary of this is that poultry manure needs to be kept dry to prevent excessive NH_3 emissions. Because of the hydrolysis of the uric acid in poultry manures in the presence of water, the solution becomes very alkaline and large NH_3 emissions occur as a consequence.

The basic processes and effects discussed in this section govern the magnitude of ammonia emissions resulting from different environmental conditions and agricultural practices. It is important to keep these processes in mind during the following discussion of the relationships between environmental factors, agricultural practice and NH_3 emissions.

2.3. ENVIRONMENTAL FACTORS AFFECTING AMMONIA EMISSIONS

2.3.1. Topography/Relief

Relief affects agricultural activities in two main ways: firstly, topography (Figure 2.2.) modifies the climatic features of the land, and secondly it influences the way any potential agricultural land can be cultivated. For instance, an increase in altitude above sea level inversely affects the ambient temperature (-0.6°C per 100 m adiabatic lapse rate) and thus limits the length of the growing season, as well as delaying the harvest. This limits the range of crops that can be grown (Grigg, 1995). Aspect and slope also affect the suitability for mechanised cultivation, frost incidence ('frost hollows'), wind exposure and hours of sunshine. In most lowland areas, the differences in relief are less important than soil and drainage differences (see 2.3.3.), however, in the upland areas of the west and north, topography is a major constraint for agriculture. Many of these areas are affected by poor drainage, shortening of the growing season and increased humidity due to orographic precipitation (see 2.3.2.). As a result, most of the higher areas are either not used or only very extensively used for agricultural purposes, with the main use being grazing on heather or grass moorland areas. The upper limit of improved land varies with economic conditions (Coppock, 1976a) including subsidies, as well as geographically with the size of the upland mass, with its latitude and the oceanicity of its climate.

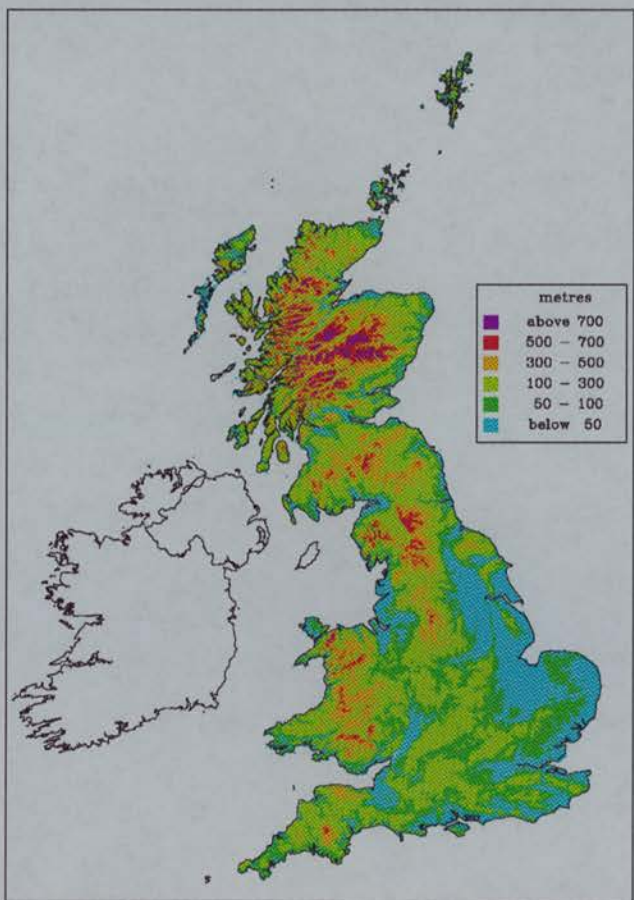


Figure 2.2. Altitude in metres above sea level (averaged at the 1-km scale); source: Countryside Information System (CIS, 1993).

2.3.2. Climate

"Diversity is the outstanding characteristic of agriculture in the British Isles. ... The range of climatic opportunity may be measured by comparing the farming in southwest Cornwall, the extreme growth conditions of which are reflected by the 'tropical gardens' of Tresco, with those of north-eastern Scotland, where the number

of frost-free days around Perth may be 70-80 less and the temperature range twice as great; or by comparing the humidity, precipitation, and cloudiness which conspire to foster ... the saturated margins of Loch Neagh, with the light precipitation and drying winds that exaggerate the aridity of East Anglia." (Mead, 1964)

The climatic factors influencing agriculture in the UK most are temperature and rainfall. Temperature is the major limiting factor for the length of the growing season and also the rate of plant growth. This is closely linked to the potential maximum available length of the grazing season. The growing season and thus the grazing season varies significantly from one part of the British Isles to another, generally getting shorter from the south to the north and from the west to the east (see Figure 2.3.). Different methods have been developed to define the length of the growing season. Some methods calculate accumulated temperatures ('T-sum 200', see: Down, 1981; Frame, 1992). Gregory (1954, 1964) and Coppock (1976a) propose a monthly mean temperature of +6°C (42.8°F) as an index of grass growing season conditions (Figure 2.3.). This limits the length of the growing season to as little as 4 months or less in the higher parts of the Scottish Highlands and the Welsh mountains. Along some western and southern coasts of Ireland, Wales and southwest England, on the other hand, the grass growing season lasts between 9-12 months. Over most of the lowlands, the growing season may last for 7-8 months, decreasing to 5-6 months over higher ground.

Low temperature, especially when accompanied by high humidity, makes it necessary to house many types of livestock. Dairy cows are in greater need of such protection in winter than the hardier and less specialised beef cattle breeds and sheep. Coppock (1976a) remarks that sample surveys by the Ministry of Agriculture and the Milk Marketing Board have shown that the length of the time dairy cows are housed varies significantly within Great Britain. While housing is required from late October for 6 months in northern England, in southern England most cattle herds stay outside at night until December. The longer the housing period is, the higher the annual NH_3 emission per animal is likely to be (see Section 2.2. and Chapter 3). For upland and hill sheep, winter temperatures, especially in combination with snowfall and/or high windspeed, determine whether supplementary feeding or even housing is necessary.

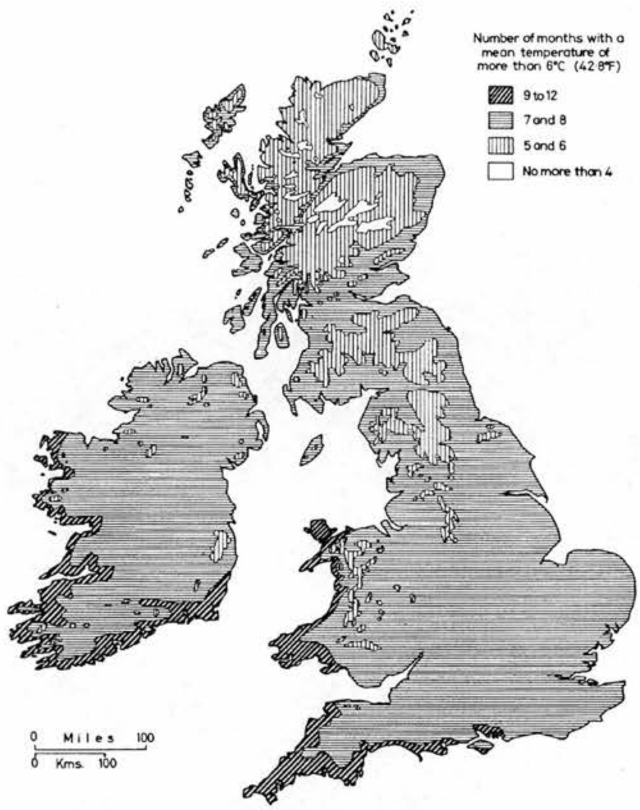


Figure 2.3. The average length of the grass growing season for the British Isles, defined by the number of months with a mean temperature above 6°C (from Gregory, 1964).

The amount and frequency of rainfall are both important in determining whether an area is more suited to arable crops or grassland. The annual average precipitation varies across the country from about 500 mm along the Thames estuary to about 5000 mm, with the highest values in some mountainous areas in the west (Figure 2.4.; Gregory, 1964). This largely reflects the prevalent direction of approaching weather systems, and is accentuated by the location of the upland areas in the western parts of the country. In general, the wetter western parts are less suited to arable cropping, but provide good grazing for large parts of the year. For instance, the heavy and frequent rainfall of Wales, the Lake District or southwest England, makes conditions for harvesting crops, particularly cereals, hazardous. However, if an area is too wet, even the quality of grass deteriorates due to insufficient drainage, poaching (destruction of the sward by trampling of animal hooves) etc (e.g. Grigg, 1995). In eastern areas, rainfall is not only lower than in the west, but also water loss through evapotranspiration is greater because of higher temperatures, which enhances the east-west contrast. Other important and limiting factors for agriculture are the seasonal distribution of precipitation and the variability between years, as long-term averages can be misleading. Grigg (1995) states that wheat yields in Britain are inversely related to summer rainfall, in contrast to grass, which is positively related. This gives an advantage to the drier southern and eastern areas for cereal growing.

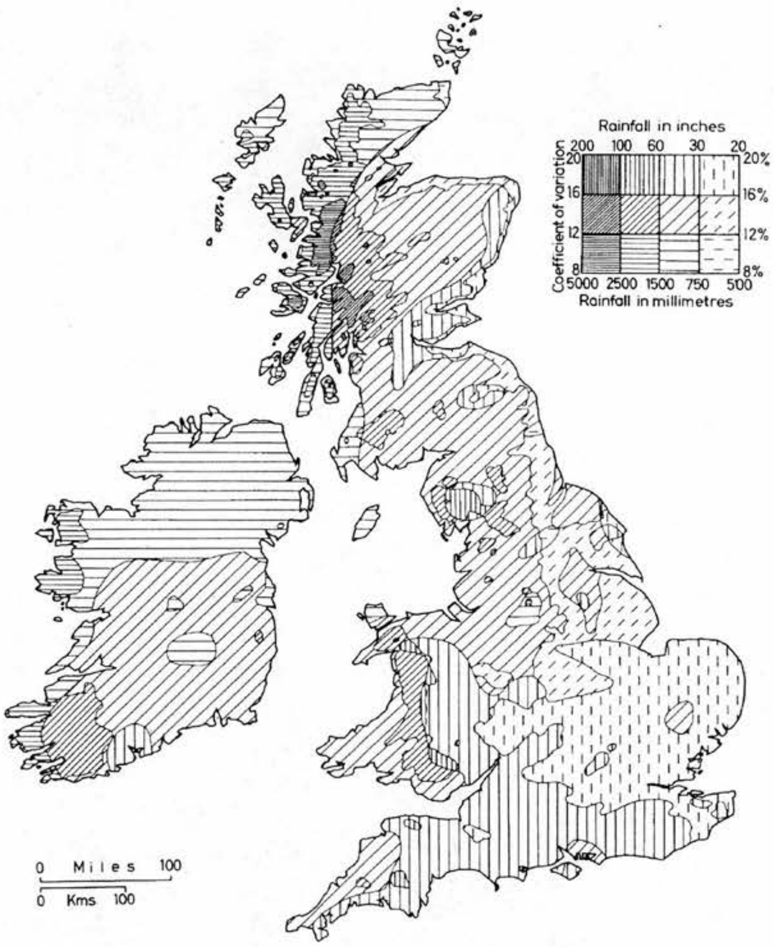


Figure 2.4. Mean annual rainfall and its coefficient of variation (from: Gregory, 1964).

These factors are of considerable importance to the farmer, especially in marginal areas. For most farming purposes, the amount and distribution of rain during summer is of most concern, although the depth and duration of snow is equally significant to hill farmers. Regarding NH_3 emissions, wetter and colder weather is less likely to cause high NH_3 emissions than warm and dry conditions (see Section 2.2.). For instance, emissions from fertilised crops are estimated to be significantly higher in warm temperate countries such as Greece or Spain, compared with cool temperate and temperate countries such as the Nordic countries or the UK (TFEI, 1996). However, secondary effects of longer winters causing higher emissions due to longer housing periods for cattle have to be taken into account when emission source strength estimates are made (see Section 10.2.).

2.3.3. Soil fertility

The suitability of any soil type for agricultural use is, to a large extent, linked to the effects of local topography, geology and climate. Soils vary considerably in their characteristics - structure, depth, texture, plant nutrient content and acidity. This affects both the range of crops that can be cultivated, and the yields that can be reached (Grigg, 1995). In order to explain regional differences in agriculture (and also in NH_3 emissions), these variations are of significant importance. However, soil conditions can be improved more easily than climate or slope. For instance, nutrient-poor soils can be fertilised with the relevant nutrients and minerals, acidity can be treated with lime, and waterlogged soils can be drained (Grigg, 1995).

Whereas light sandy soils warm up quickly and are more easily cultivated, they are also generally lower in nutrients and more prone to droughts due to their porous texture (Coppock, 1976a; Grigg, 1995). Heavy, fine-textured clay soils, on the other hand, are less easy to work, as their water-retaining properties make them more susceptible to damage by poaching and heavy machines (Frame, 1992; Grigg, 1995). They are generally better suited for grass and thus for cattle and sheep farming, especially in the less wet areas of the country, whereas grass on light soil under dry conditions is likely to burn (Coppock, 1976a).

It is therefore difficult to assess the agricultural value of any particular soil type without looking at the climate and relief at the same time, i.e. the soil type cannot be used in isolation to determine the suitability of an area for any particular crop (arable or pasture). This leads to similar soil types having a different suitability for any particular crop, depending on which part of the country they are found in. For instance, soil ideally suited for grass is not necessarily the best soil for arable crops. Also, a soil which is too light with very high drainage in the drier east, may be ideal in the high precipitation areas of the west. In order to provide a more useful view of the suitability of different soils for agricultural purposes, land classifications, land capability maps (Figure 2.5) etc. have been produced, which also take climatic conditions and the relief into account (Hogg, 1962, Bibby and Mackney, 1969; Bibby, 1982; House, 1982; Bibby and Thomas, 1990).

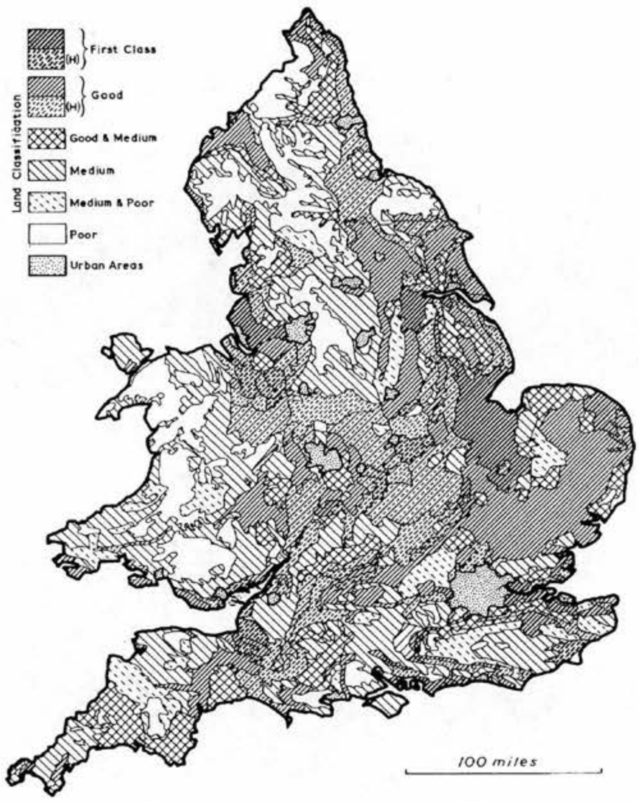


Figure 2.5. Land classification for England and Wales (from: Coppock, 1976a).

As mentioned above, the natural condition of most soils can be enhanced by improved drainage (in areas of high precipitation) and fertiliser application. Nitrogen (N), phosphorus (P) and potassium (K) are the most important nutrients influencing grass or crop growth, and their supply can be influenced to a large degree by the farmer. Economic reasons have lead to an increased use of fertilisers, especially N fertilisers, to aid the intensification of agriculture. Without N fertilisers, most grass swards produce 2-5 tonnes (t) dry matter (DM) per hectare, depending on the general soil quality and the amount of N available in the soil (Frame, 1992). With large N fertiliser application rates, dry matter production may rise to above 10 t DM per hectare in practice, with potential production rates of above 20 t DM on experimental plots under ideal conditions. For both crops and grassland, higher application rates of N fertilisers increase the potential for NH_3 volatilisation (e.g. Holtan-Hartwig and Bøckman, 1994). Any increase in the N content of livestock feed, i.e. grazed grassland, grain or fodder crops (such as maize, turnips etc.), is translated into higher

NH_3 emissions (e.g. Jarvis and Pain, 1990; Jarvis *et al.*, 1989b; Orr *et al.*, 1995). Aspects of farming practice relevant to NH_3 emissions are discussed in Sections 2.4. and 2.5. regarding livestock farming and fertiliser N application as the main NH_3 sources. Ammonia emission source strength estimates from the literature are examined more closely in Chapter 3.

2.4. LIVESTOCK FARMING PRACTICES AFFECTING AMMONIA EMISSIONS

The volatilisation of ammonia from livestock manures is the largest NH_3 source in the UK (e.g. Sutton *et al.*, 1995; TFEI, 1996; Pain *et al.*, 1998) . It is therefore important to obtain an overview over the different livestock farming practices and the factors influencing NH_3 emission potential. This is essential for understanding the influence of differences in farming practice as they are relevant to NH_3 emission and the uncertainties associated with their estimation.

Livestock farming in the UK is very diverse, ranging from intensive dairy farming to extensive hill sheep farming, both of which are associated with grassland, to intensive poultry and pig farming, which is less closely related with the land use in their locality.

2.4.1. Livestock types and breeds

Over many thousand years, animals have been bred to develop certain characteristics, which makes them especially suited to certain purposes (Figure 2.6.). An example of this is the differentiation between dairy and beef cattle. Whereas dairy cattle breeds (such as Friesians, Jerseys or Guernseys) are highly specialised to maximise milk yield and milk quality, beef cattle (e.g. Charolais, Aberdeen Angus) are bred to produce high quality meat. There are large differences in the external appearance and hardiness of different cattle breeds. Examples for this wide variety are the small and more sensitive Channel Island breeds (380 kg liveweight Jerseys; 430 kg Guernseys), the very large beef breeds such as Charolais (685 kg), or the hardy and more "all-round" hill cattle breeds (Highland Cattle, 495kg) which produce much less milk and have lower feed requirements (MAFF, 1980a). The same degree of variation applies to sheep breeds, the specialisations being wool, meat or a combination of both. In the UK, sheep production is geared essentially to meat production, with wool mostly as a useful by-product (MAFF, 1987c). Different breeds range from very small and hardy hill sheep (such as Welsh Mountain Sheep; 32-50 kg liveweight) to mediumweight (51-68 kg; e.g. Finn Dorset) to the heavyweight lowland sheep breeds (69-91kg; e.g. Merino sheep) (MAFF, 1980a).

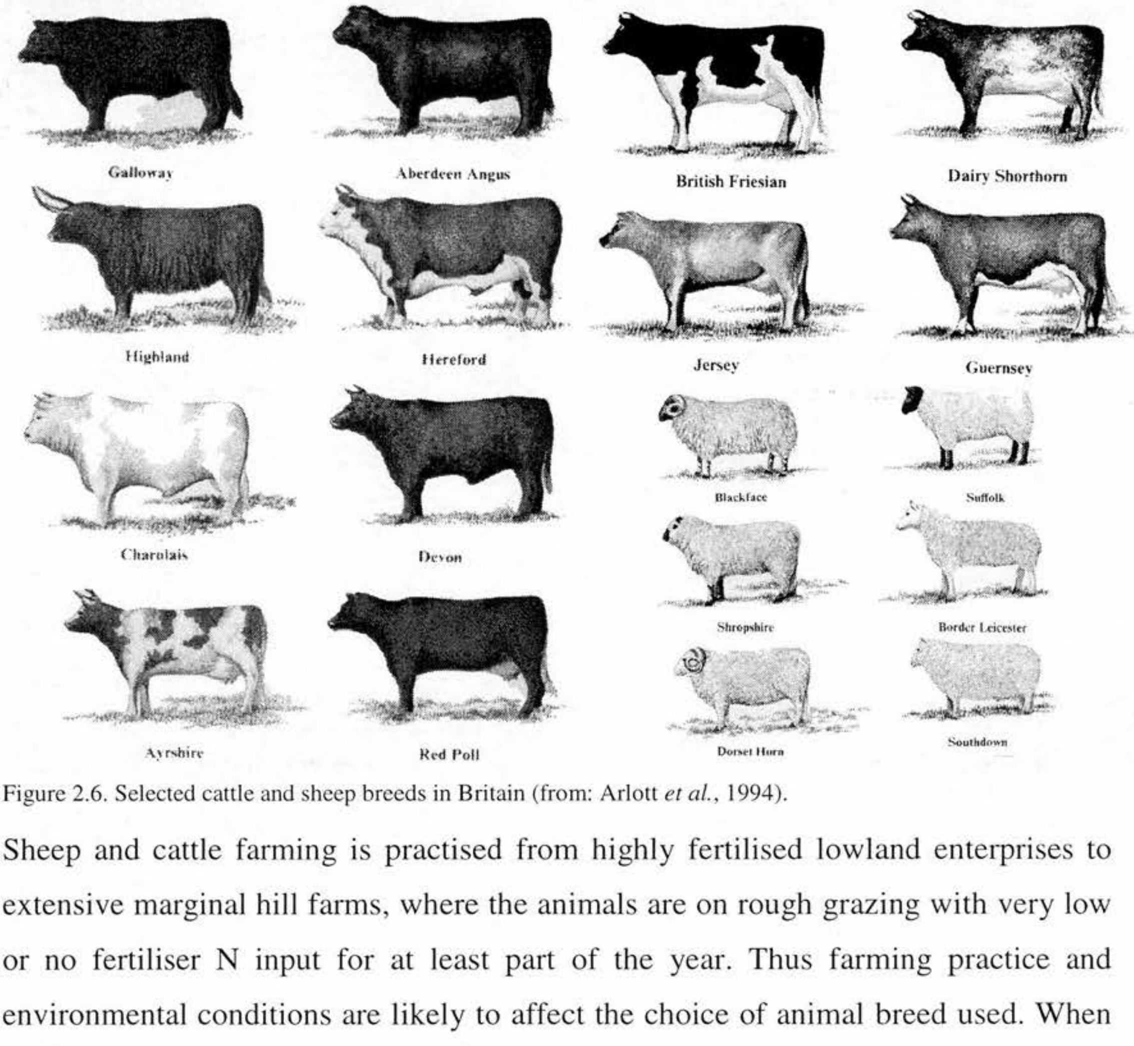


Figure 2.6. Selected cattle and sheep breeds in Britain (from: Arlott *et al.*, 1994).

Sheep and cattle farming is practised from highly fertilised lowland enterprises to extensive marginal hill farms, where the animals are on rough grazing with very low or no fertiliser N input for at least part of the year. Thus farming practice and environmental conditions are likely to affect the choice of animal breed used. When assessing NH_3 emissions, the difference in size, nutritional and housing requirements of the breeds is an uncertainty factor for NH_3 emission models, as the models tend to represent each livestock type as an average animal. The key differences between different livestock breeds regarding NH_3 emissions are due to animal size and the grazing and housing conditions (feed N content, housing duration etc.), which determine N excretion rates and NH_3 source strength. It is generally assumed that this does not pose a too large problem in non-spatial inventories, as the differences should even out. However, in spatially distributed inventories, it makes a difference whether large herds of lowland sheep or the much smaller hill sheep are grazing in an area (see also Section 6.2.2.). Also, the spatial distribution of average cattle animal emissions is significantly different from a separate distribution of beef and dairy cattle, especially with regions predominantly specialised in either dairy OR beef in the UK (Chapter 9).

2.4.2. Grassland and grazing systems

In his *Natural History of Selborne* (1788), Gilbert White called the British Isles 'a grazing kingdom'. Grassland comprises about 70% of the UK's agricultural land area (Figure 2.7.). Grass is the primary feed source for ruminant livestock and provides about 60-65% of the diet for dairy cows, 80-85% for beef cattle and 90-95% for sheep (Frame, 1992). It is the most economic crop for the greater part of the UK, with a large number of different grassland types and management techniques, such as different grazing systems, hay making or ensiling (Mead, 1964).

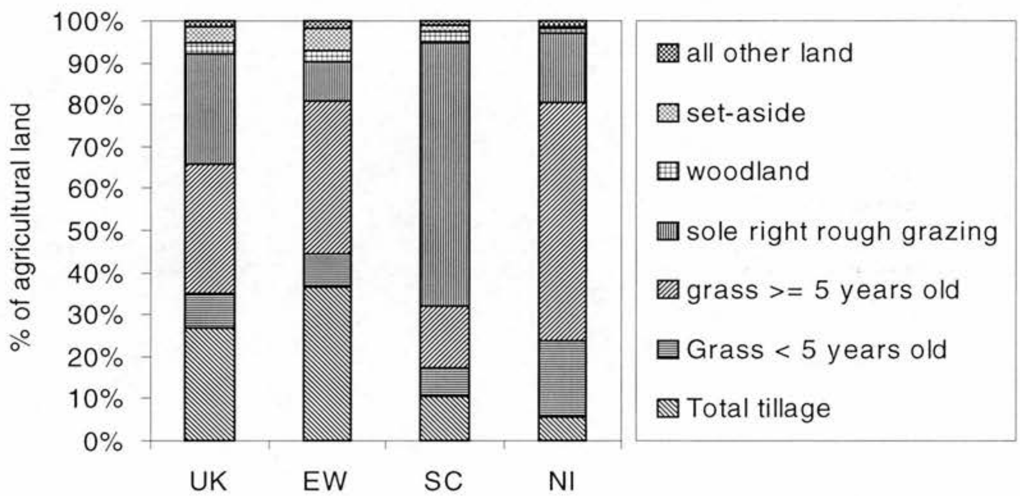


Figure 2.7. Use of agricultural land in the UK, England & Wales, Scotland and Northern Ireland (after GSS, 1996).

Fields may be used continuously for grass growing (permanent grass), or alternated with arable crops (ley grass). Leys may be short (1-2 years) or long term. The Agricultural Census (e.g. GSS, 1996) distinguishes between swards under five years old and older grass (five or more years). The grass over 5 years old constitutes the majority in Britain. It may be either in long term rotation (e.g. 10 years or more) with arable crops or it may be permanent grassland. In general, well established and managed permanent grass swards may be as productive as or more productive than short term grassland.

Frame (1992) distinguishes four main types of grassland for Britain and many European countries. Regarding NH_3 emissions, only the first three categories are relevant:

- specialist intensively managed grassland with relatively high fertiliser N input and high stocking rates, e.g. for dairy herds grazing;
- less intensively stocked grassland, based on temporary grass and/or forage legume swards receiving less to very little or no fertiliser N, or on long term to permanent grass receiving moderate inputs of fertiliser N, e.g. for lowland sheep and beef cattle grazing;
- extensively used grassland, e.g. for hill sheep and suckler cow grazing;
- grassland, primarily in the hills and uplands, diverted to non-agricultural uses, e.g. national parks, wildernesses, leisure and recreation areas.

As mentioned earlier, the area of agricultural land under grass generally increases from east to west and from southeast to northwest, as a result of increasing rainfall, decreasing soil fertility and the various difficulties with arable cropping. Grassland also becomes more prevalent at higher altitudes, with an increase of rough grazing and a shorter growing season (see 2.3.) due to lower temperatures and poor soil conditions. It should be noted that the grazing season is about 5-6 weeks shorter than the growing season, since there has to be sufficient grass on offer before grazing can begin (Frame, 1992). The potential for livestock grazing also becomes more limited with the severe restrictions on the use of improvement techniques imposed by climate, soil and topography. As the conditions become more extreme, the economies of fertiliser application, drainage, reseeding etc. become less viable.

The effect of DM response to fertiliser application has been studied intensively (Frame, 1992; Scholefield *et al.*, 1991; MAFF, 1982c). Figure 2.8. and Table 2.1 show average yields of grass for a range of different fertiliser application rates over the spectrum of soil quality ('site class'). It illustrates how the better soils will give optimum yields at a much higher fertiliser N input than poorer soils.

Table 2.1. Probable dry matter yields (t ha^{-1}) over a range of fertiliser N application rates (kg N ha^{-1}) for each of 5 site classes (MAFF, 1982c):

Site class	0 kg	50 kg	100 kg	150 kg	200 kg	250 kg	300 kg	350 kg	400 kg	450 kg
Poor	1.6	2.9	4.2	5.4	6.6	7.7	(8.4)			
Fair	2.0	3.3	4.6	5.9	7.2	8.2	9.1	(9.5) [a]		
Avg.	2.4	3.8	5.1	6.4	7.7	8.7	9.6	10.3	(10.5)[b]	
Good	2.8	4.2	5.6	7.0	8.3	9.4	10.3	11.0	11.5	(11.6)[c]
Very good	3.2	4.7	6.1	7.5	8.9	10.0	10.9	11.6	12.2	(12.7)

( ) = optimum yields at optimum fertiliser levels, a =at 330 kg ha^{-1}, b = at 370 kg ha^{-1}, c = at 410 kg ha^{-1}.These DM yields assume an average available soil N level (soil index 1).

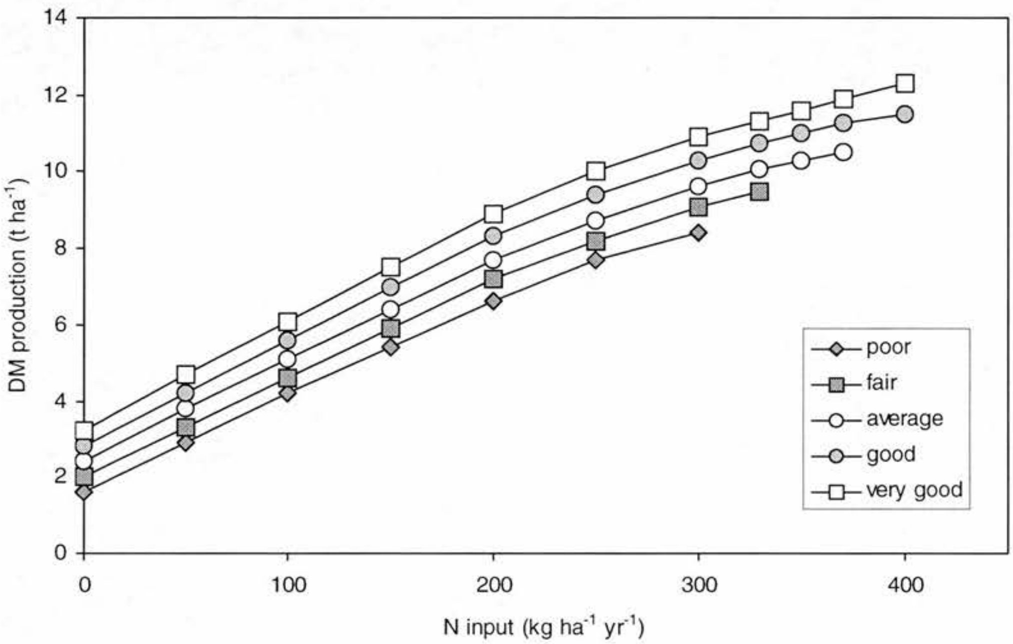


Figure 2.8. Dry matter production responses to fertiliser N input (derived from Table 2.1. above; after MAFF, 1982c).

The essential elements of any grazing system are the total amount of grass produced and the seasonal pattern of grass growth (MAFF, 1982a, b, c). When the N fertiliser application rates are increased on a grass sward, the herbage production (15-25 kg DM per kg N response) increases more or less linearly up to N rates of 250 to 350 kg ha^{-1}. The amount of response and the fertiliser rate needed to achieve the response are influenced by soil characteristics, moisture supply and also the frequency of defoliation (through cutting or grazing). If the annual N application rates are increased further to 450 kg ha^{-1}, the herbage DM response for each added kg N decreases (to 5-15 kg DM per kg N), following a law of diminishing returns. It is important to realise that between 80 and 90% of the maximum DM production can be achieved with 50 to 60% of the maximum N application rates. Each farmer has to decide, depending on the type of enterprise, which N rate is economic, i.e. a herbage

DM response of 5-10 kg DM per kg N applied may be viable for a dairy farm, but not for a beef or sheep farm with its lower output per kg DM (Frame, 1992).

It should be noted that forage legumes such as clover, lucerne or sainfoin as well as peas and beans have the ability to fix N through bacteria living symbiotically in root nodules (rhizobial N fixation). They also improve the soil condition in general and provide good forage for animal production. The growing of legumes in rotation with cereals and root crops has been an important traditional method of maintaining the N level in Western Europe in order to sustain crop yields (Grigg, 1995).

The amount of N fixation in grass/clover swards varies widely depending on the clover content, but ranges of 75-280 kg N ha^{-1} in lowland swards and 100-150 kg N ha^{-1} on hill and upland pastures have been reported from experiments (Frame, 1992). The amount of fixed N is potentially comparable to fertiliser N application rates. However, high N application rates in grass/clover swards reduce the clover content in swards and depress the N-fixing capability of the rhizobia, i.e. the lower the fertiliser input, the higher the N fixation through legumes. These advantages of N fixation regarding large potential savings for the farmers, have been neglected in Europe in recent times, during an era of heavy fertiliser N usage. With new agricultural policies there is a growing interest in the potential of forage legumes (Frame, 1992). For instance, N fixed by clover may reduce the run-off and leaching rates, while providing an equivalent amount of N to the sward. Regarding NH_3 emissions, however, cattle or sheep grazing on a grass/clover sward with N fixation rates of 200 kg ha^{-1} produce a similar amount as on a field fertilised with 200 kg N input as mineral fertiliser (e.g. Jarvis and Pain, 1990; Orr *et al.*, 1995).

The higher the fertiliser application or clover content, the higher the potential carrying capacity for grazing animals becomes. However, the rate of grass growth and therefore the grazing stocking rate varies considerably during the grass-growing season (Figure 2.9.). During March to May/June the growth rate increases to a maximum around early June, decreases until the end of July and increases to a secondary, but much smaller peak until early September and then it decreases again until mid-November. Winter growth is minimal. This pattern of a peak of growth in spring and a smaller peak in autumn is not consistent from year to year and differs in emphasis between the drier and wetter areas of the country (MAFF, 1982 a, b). The seasonal variation in grass growth can be modified only to a certain extent by the use of different grass strains, the timing of N applications and irrigation.

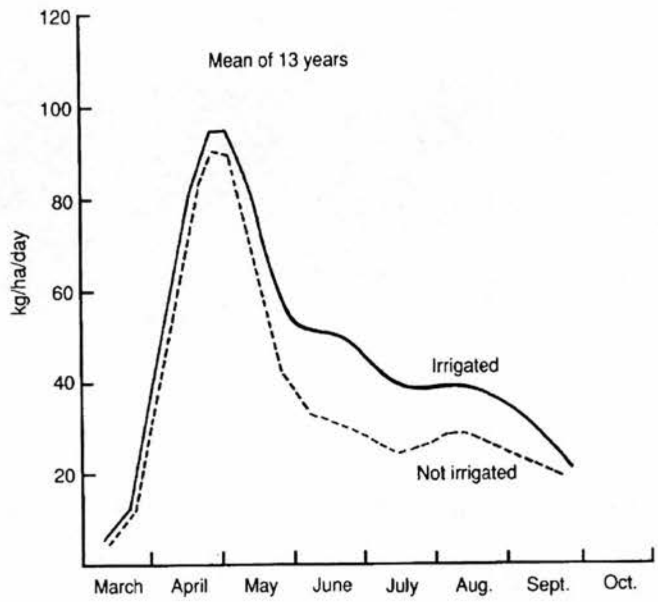


Figure 2.9. Seasonal grass growth rates (from Frame, 1992).

The DM production from an area not needed for grazing in the early part of the year, when the highest grass growth rates occur, can be conserved as silage or hay for times when supplementary feeding of grazing livestock is necessary. The transformation of grass to silage occurs through controlled fermentation primarily by lactic acid bacteria under anaerobic conditions. Currently over two thirds of all conserved grass is silage, a reversal of the situation prior the late 1970s, when hay was the principal method of crop conservation in the UK (Frame *et al.*, 1995), due to new techniques developed for more successful ensiling. Silage making is less weather dependent than hay making.

Although grass is the principal crop for silage, the ensiling of maize and whole-crop cereals has increased in recent years. Highly fertilised grass has a higher moisture content than grass with less N input. This makes it less suitable for haymaking, but more suitable for ensiling. As grass can be ensiled at an earlier stage of growth (when it has a higher moisture content), the total annual DM production increases, because the grass can be cut more frequently. Thus the farmer can take advantage of the rapid growth phase in spring, take a first cut, and then get another cut (up to 3-5) before the grass growth slows down. The higher fertiliser content in grass grown for ensiling results in a higher N content in the silage fed to livestock, which in turn results in higher NH_3 emissions. It follows that, on average, animals fed on hay during the housing period emit less NH_3 due to their lower N diet.

Rough grazing is a generic name for semi-natural plant communities which provide extensive grazing, mainly on open ranges with a short growing season. Unfertilised bent/fescue grass swards produce 1.0-3.6 t DM per hectare during the growing season, with production rising to 3.5-4.5 t under moderate fertilisation. Grass or shrub heaths provide 1-2 t ha^{-1} DM, young heather 1.5-3.0 t ha^{-1} (Frame, 1992; MAFF, 1991a; see Table 2.2.).

Table 2.2. Levels of herbage dry matter (DM) production typical for hill vegetation (MAFF, 1991a).

Vegetation type	kg DM ha^{-1}
Improved grass	5000
Good native grass	3500
Poor native grass	2000
Young heather (up to 20cm height)	1750
Intermediate heather (20-35/40cm height)	1750
Old heather (mature & degenerate > 40cm)	1250
Blanket bog	1400

Heather moorland is normally made up of a mosaic of vegetation types. Grassy communities are common on lower slopes, and blanket bog mixed with heath, sedge and grass species occurs on wetter or less freely draining areas. Sheep normally have access to graze the whole range and are able to exercise choice within and between different vegetation types. Sheep prefer to graze on (MAFF, 1991a):

- herbs (like clover, yarrow etc.) rather than grasses
- broad-leaved grasses in preference to fine-leafed grasses
- live leaf rather than stem or dead leaf
- grasses and sedges rather than heather in heather or bog communities

Generally better quality bent and fescue grasses (*Agrostis* and *Festuca*) are grazed heavily all year round. Where *Agrostis/Festuca* are in short supply, the poorer grasses attract heavier grazing pressures. Heather, which has a very low feed value, is grazed mainly in winter, when the amount of green herbage in the grass communities is low. The grazing pressure on heather is influenced by the amount and quality of grass in the neighbourhood as well as by the condition of the heather itself, which is managed by periodic burning. The primary aim of heather burning is to remove ageing stands and restore the vegetation to a condition where young growth is more accessible to grazing livestock, as old heather progressively loses its ability to regenerate. Poor burning practice causes localised overgrazing and can result in the gradual decline and disappearance of the heather. This in turn results in a reduced carrying capacity.

Regarding NH_3 emissions, rough grassland, grass moorland and heather with no or very low fertiliser N input provide a diet with relatively low N content to the grazing livestock. This results in lower NH_3 emissions per animal than on highly fertilised grass swards in the lowlands (e.g. Orr *et al.*, 1995). The sparsity of dry matter provided by these rough pastures also determine lower carrying capacities, which result in the low emission rates per unit area from the thinly spread grazing livestock, compared with rich pastures in the lowlands.

2.4.3. Carrying capacity of grazing systems

Carrying capacity is generally expressed as the number animals of a certain type (e.g. dairy cows, lowland ewes) per unit area as an annual average (MAFF, 1980a). Clearly, there are large differences in the intensity of grazing possible on highly fertilised lowland grass swards and hill pastures. The effective carrying capacity is thus not only influenced by animal numbers, but also by the size/age of the animals and, for sheep, lambing percentage (MAFF, 1983a).

As stocking rates increase, so does the total animal production per hectare, but normally, after a certain threshold (max. stocking rate), the production per animal begins to decrease. A sustainable stocking rate is influenced by the amount of grass grown which in turn is influenced mainly by the amount of N fertiliser applied (MAFF, 1982a). Some examples of sheep carrying capacities for different types of pasture are shown in Table 2.3.

Table 2.3. Carrying capacity of an optimally fertilised pasture (see also Table 2.1. and Figure 2.8.) for lowland sheep (from: MAFF, 1983a).

Grass growth class	Potential yield (t ha^{-1})	Carrying capacity (70 kg ewes + 1.8 lambs)
Poor	8.4	9.0
Fair	9.5	10.2
Average	10.5	11.3
Good	11.6	12.5
Very good	12.7	13.7

Studies of different rough grazing communities have shown carrying capacities for sheep to vary between 0.7 and 2.4 ewes per hectare, depending on the vegetation composition (MAFF, 1991a). Hill and upland grazing all year-round can only sustain a relatively low animal production. In recent years, however, large increases in sheep output have been obtained on hill farms due to the introduction of the 'two-pasture system'. This is mainly based on strategical use of relatively small enhanced areas at critical stages in the breeding cycle, in conjunction with rough or unimproved hill grazings (MAFF, 1981a; MAFF, 1990). These improved areas close to the farm steading are also called 'in-bye'.

2.4.4. The grassland N cycle

The N content of grass is influenced by various factors, but mainly by the amount of N fertilisers applied. A large proportion of the nutrients ingested by grazing livestock (between 75 and 90%, depending on the type and class of animal) is excreted as dung and urine. Excretion off the pasture, leaching and volatilisation of N from urine and dung patches are sources of loss from the sward N cycle. The role of the excreta of grazing animals in the N cycle and also in environmental pollution has been recognised for some time. There are major differences between dung and urine in N content and in the amounts and availability for plant growth, as well as regarding NH_3 emissions (Ryden *et al.*, 1987).

Dung mainly consists of undigested cellulose and lignin residues, waste mineral matter and living or dead ruminant micro-organisms together with their metabolic products. The water content varies between 85% for cattle dung and 65% for sheep dung. Nitrogen in dung is largely contained in organic compounds, which is released very slowly through decomposition by micro-organisms (Ryden *et al.*, 1987; Frame, 1992). Thus, NH_3 emission rates from dung on pastures are very small compared with urine (e.g. Ryden *et al.*, 1987; Jarvis *et al.*, 1989a).

Urine contains over 90% water, as well as nitrogenous compounds (mostly urea) from the breakdown of protein. The proportion of excreted N in urine increases with increasing N content in the diet, almost all of which is readily available. Because of the rapid hydrolysis of urea and the high local pH, a significant proportion of the available N is lost by volatilisation of NH_3 (see also Section 2.2.). Weather conditions play an important role in the processes of N removal, increasing volatilisation under hot, dry conditions, while rainfall causes leaching of urea and nitrates from NH_3 nitrification (Ryden *et al.*, 1987; Jarvis and Pain, 1990; Frame, 1992).

The better the nutritional value of the sward (i.e. the higher the fertiliser application to grassland), the higher the carrying capacity of the sward becomes (see Section 2.4.3. above). In consequence, this increases the NH_3 emissions per animal, due to the higher N content of the grass, as well as the total NH_3 emissions per unit area, due to the higher density of animals (Jarvis and Pain, 1990). Grazed grassland may also receive N from animal manures collected through the housing period. This is discussed further in Section 2.4.6.

The complete N cycle (Figure 2.10.) for grazed grassland can be described as follows: the main components are N inputs to and N losses from the system as well as recycled N. Nitrogen inputs originate mainly from the application of organic manures, mineral N fertilisers, symbiotic N fixation through legumes, mineralisation from soil organic matter and wet and dry deposition from the atmosphere. Desired outputs are animal products such as milk, wool and meat, conserved silage or hay. Less desired outputs are losses of NH_3 through volatilisation from urine, leaching of nitrate through drainage, denitrification of nitrate to N_2 and N_2O, or run-off of slurry. Nitrogen recycling in the system occurs via a number of pathways which include not utilised herbage and root tissues being broken down through senescence and soil organisms, nitrification of ammonium to nitrate by bacteria in the soil, and the breakdown of urine and dung on the sward.

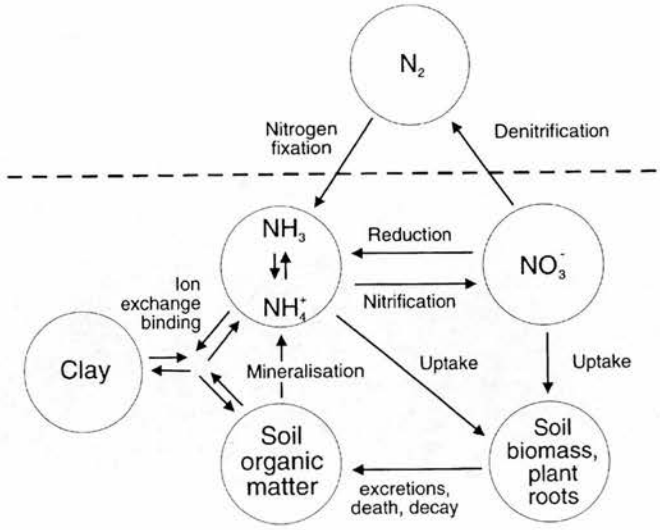


Figure 2.10. Simplified nitrogen cycle (from ECETOC, 1994).

Historically, animal manures were not regarded so much as a waste product as they are today, but as a valuable resource of plant nutrients. As Potts (1807) stated, dung is a '*universal fertiliser. ... The grand property of dung is therefore to yield immediate food to plants. ... Its effects have powerful progressive influences.*'

Inorganic fertilisers have been widely used in traditional farming systems, e.g. bone meal to supply phosphorus. The modern fertiliser industry dates back to the 1840s (Grigg, 1995), but only after the Second World War fertiliser became sufficiently cheap for intensive applications. Over the last few decades, changes in livestock farming (intensification, especially of pig and poultry rearing; housing changes) have resulted in large quantities of manure being produced, with not much suitable land for manure spreading close by. For instance, pig and poultry numbers have risen by a factor of 3 and 5, respectively, between 1870 and 1995 (Figure 2.11.). Cattle and sheep numbers in the UK have increased by a factor of 2 and 1.5. respectively, over the same period.

These changes, together with the wide availability and low cost of mineral fertilisers resulted in manure being regarded very much as a disposal problem rather than a source of nutrients (Frame *et al.*, 1995). Dampney and Unwin (1993) estimate that a typical 100-cow dairy herd produces nitrogen, phosphorus and potassium worth over £2,700 during a six months winter housing period. Many farmers continue to disregard the potential value of their manures and little allowance is made for the nutrients contained in them. This attitude is, however, changing slowly again, through increased emphasis on environmental protection (from run-off, leaching, NH_3 volatilisation and odours) and the lowering of farm input costs, promoting and resulting in a more positive and rational use of slurries and FYM (Van der Meer *et al.*, 1987). Including the nutrient content of organic manures applied to the fields as part of the total fertiliser input, rather than treating manures as additional input, can result in reduced total NH_3 emissions, due to the reduced application rates of mineral fertilisers.

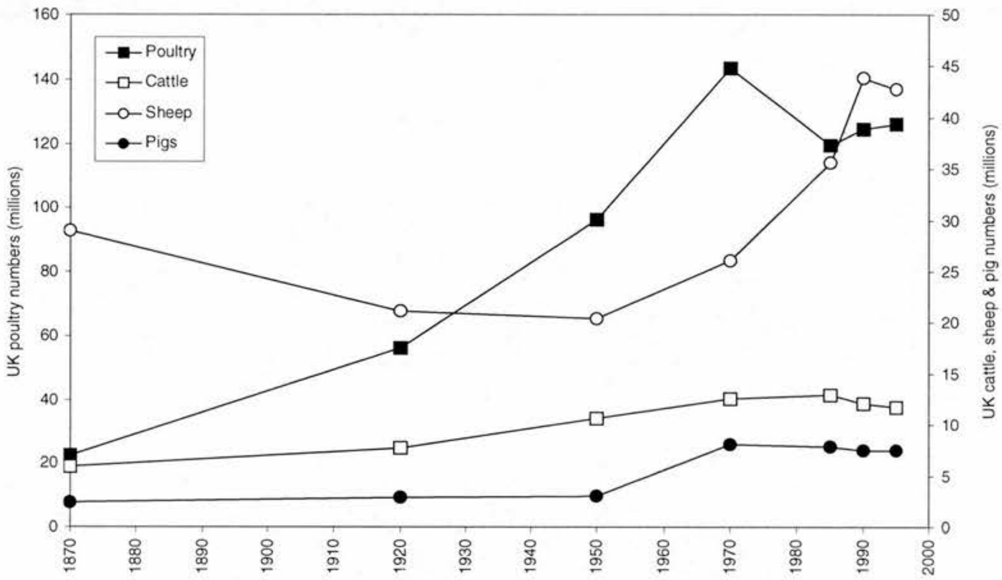


Figure 2.11. Changes in UK livestock numbers 1870-1995 (data for the period 1870-1950 from Asman *et al.*, 1988; data for the period 1970-1995 from GSS, 1973; GSS, 1993; GSS, 1996).

Abatement measures aimed at minimising N losses need to take account of the details of the N cycle. Measures designed to minimise losses through NH_3 volatilisation may well increase the rate of nitrate leaching and vice versa. There are, however, measures which could contribute to a reduction of overall N losses. Examples of this type of integrated abatement strategy are more precision in the rate and timing of fertiliser N application, the preference of certain types of fertiliser (see 2.5.), or the use of legumes where suitable.

2.4.5. Livestock housing and manure storage

Some types of farm livestock, such as cattle, sheep, goats, horses etc. spend a considerable part of the year outdoors and are housed during the winter season as well as sometimes overnight at other times of the year. Pigs and poultry are largely reared indoors, with some exceptions such as free-range pigs or poultry being outdoors for at least some part of their lives. During the housing period, the animals excrete large amounts of dung and urine, which are stored on the farm, either directly in or underneath the animal houses, or outside. Farmers have been aware of nutrient losses from storing manures for a long time. Potts (1807) recommends landspreading of manures without prolonged storage periods, because '*muck wastes by keeping to an unprofitable degree*'. He attributes this to the fact that '*the more dunghills are stirred and turned over, and rotted, the more of their virtue was lost*', thus recognising what is today known as N losses through NH_3 volatilisation. The magnitude of NH_3 emissions arising from livestock housing and manure storage varies greatly depending on the type of animal, the feed composition, the housing practice (including type of buildings and manure collection facilities), the length of the housing period and storage facilities. Climatic conditions such as temperature also play a considerable role (see Section 2.2.).

"Manures" are defined as a mixture of dung and urine in any form, i.e. slurries or farmyard manures. "Slurry" is mixed dung and urine as voided by the animal, whereas "farmyard manure" (FYM) is slurry mixed with straw or other absorbent bedding material (such as peat or sawdust) to yield a solid manure (MAFF, 1983b). The type of livestock housing will determine whether bedding materials are used, and if so, the amount. This affects the dry matter content of the manure and its handling characteristics. FYM is often stored in a heap to rot down before use (MAFF 1983b, 1976), and through the decomposition process a higher proportion of the N becomes available to plants.

Livestock housing

Livestock housing involves the production of large volumes of slurry, requiring a planned system for collection, storage and disposal (MAFF, 1987b). Traditional housing of farm livestock produces FYM, which is stackable: the animals are housed on a solid floor, and the dung is either covered with litter and allowed to accumulate (deep pit, bedded courts), or removed frequently. The latter method results in FYM heaps, which are usually stored on a concrete or other impermeable floor, providing drainage facilities for effluents, and protected from rain, to avoid leaching. In order to minimise losses, especially through NH_3 volatilisation, the heap should not be disturbed until the manure is applied to the land. Losses from housing are increased if the walls and floors of livestock buildings are constantly covered with layers of faeces or urine. The depth is less important than the surface area (Muck and Steenhuis, 1982), and removing slurry frequently by flushing, or scraping floors can help control NH_3 emissions from cattle and pig buildings (Kroodsma *et al.*, 1993).

Manure storage

Intensive pig, poultry and beef units normally have associated in-situ storage systems for slurry. These may be located under part of or the whole of the housing area, consisting of slatted floors, through which the manure passes into the storage area. This store may retain the manure for several months, and in the case of poultry on deep pit systems for the whole production cycle. Alternatively, slurries may be stored above-ground, in circular slurry stores or lagoons. These stores can be open or covered on top, the latter being recommended to avoid N losses through NH_3 emissions before application to the land (Sommer *et al.*, 1992; Olesen and Sommer, 1993). Ammonium N is lost from any slurry surface as NH_3, i.e. it is not possible to retain the full N content of fresh slurry during storage. The larger the surface of the storage area, the more NH_3 is likely to volatilise. Estimates of the N loss from slurry stores vary between 10 and 90%, depending on whether this is mainly a gaseous loss or both a gaseous and liquid loss from the store (MAFF, 1983b). Any loss of N reduces the potential value of the manure as fertiliser.

Dairy cows

Dairy cows in the UK are normally housed in cubicle systems or in loose housing, based on FYM, for about half of the year on average (Pain *et al.*, 1998). However, just as the grazing season is variable across the country, so is the housing and associated winter feeding period. Low temperatures and wet weather can delay the date when stock are turned out onto pasture, and similar weather in autumn can advance the date of housing (e.g. Grigg, 1995; Frame, 1992). The recognised housing period for dairy cows can vary from 105-235 days in the UK (Frame, 1992). As emissions tend to be much larger per housed animal per day than per grazing animal per day (see Section 2.2.), any substantial variations in the duration of the housing period cause substantial variations in the annual NH_3 emissions per animal (see Chapters 3 and 10). In addition to the winter housing period, dairy cows are normally housed for a short time every day during the grazing period for milking. After the cows have returned to their pastures, any slurry left in the milking sheds continues to emit NH_3, thus providing a large additional NH_3 source.

Beef cattle

Beef cattle are generally housed during winter for a similar period as dairy cows, although some hardy hill cattle breeds can stay outdoors for most of the year with shelter and supplementary feeding provided. Housing systems for beef cattle are loose housing in traditional FYM based cattle yards, or loose boxes or calf pens. The winter feeding of dairy cows and beef cattle is usually based on conserved grass, either as silage or hay, with straw, non-grass silage, brewer's grains, sugar beet pulp, and/or molasses as optionally added components.

Sheep

Sheep spend most of the year outdoors, with shelters against bad weather and, during winter, supplementary winter feeding on restricted pastures closer to the farms and stubble turnip fields. Sheep are grazed on turnips in two stages from late autumn onwards, the green leaves first and then the turnips themselves straight from the soil. Supplementary feeding usually starts at least 6-8 weeks before lambing and lasts until some time after lambing (MAFF, 1983c). The housing period normally does not start until January, weather permitting, and lasts on average only about one month in Britain for lowland sheep (Pain *et al.*, 1998). In-wintering of flocks has become more popular for a number of reasons, mainly to assist intensification. This benefits the sheep by ensuring a higher survival rate of ewes and lambs, and the pastures, as winter-rested pastures allow increased overall stocking rates and earlier growth of spring grass. It has therefore become a widely accepted practice in intensive sheep production systems (MAFF, 1981b, 1987c).

Pigs

Pigs are traditionally housed all year round on straw-based manure or in slurry-based systems on slatted floors with minimal straw provisions. Depending on the feeding regime (dry meal or liquid based diets), pig manures may contain large amounts of liquid.

Poultry

Poultry manure, in contrast, is largely dry, with deep pit or FYM-based deep litter systems. In battery houses the manure is collected on belts between the cages and removed to a store. The drier poultry manure is kept (sometimes with the help of additional air drying facilities), the less offensive the smells and also the less N is lost as NH_3 to the atmosphere. This is because uric acid, the main N component in poultry manure, causes large NH_3 emissions through hydrolysis when in contact with water (see Section 2.2.).

As mentioned above, housed livestock under different feeding regimes produce different amounts and consistencies of slurry. Typical values are shown in Table 2.4.

Table 2.4. Average quantities and characteristics of manures produced by livestock (from: Pain *et al.*, 1998).

Livestock type	Slurry output (t animal^{-1} year^{-1})	FYM output (t animal^{-1} year^{-1})	TAN content slurry (kg m^3)	TAN content FYM (kg m^3)
Dairy cow	10.8	14.0	2.25	0.6
Other cattle > 2yrs.	5.8	7.5	1.75	0.6
Other cattle 1-2 yr.	4.7	6.1	1.75	0.6
Mature sheep	1.5	2.0	-	0.6
Breeding sow	4	5	4.2	0.7
Fattening pig > 110 kg	2.0	2.2	4.2	0.7
Fattening pig 20-110 kg	1.5	1.6	4.2	0.7
Fattening pig < 20 kg	0.5	0.5	4.2	0.7
1000 laying hens	-	42.0	-	9
1000 broilers	-	27.0	-	12.4
1000 turkeys	-	42.5	-	12.4

The storage of livestock manure forms an integral part of the manure management system and should be designed to cover the period when conditions are likely to prohibit application to the land. The storage period typically varies from 12 to 30 weeks (MAFF, 1984), depending on the following factors:

- Location of the holding including distance from urban areas and topography of the site
- Sensitivity of the site's water resources to pollution and climatic conditions such as the length of dry periods, and the intensity and volume of rainfall at critical times of the year (to reduce the risk of leaching);

- Soil type: there is a wide range from free draining to heavy clay; this affects the capacity to use heavy wheeled vehicles for landspreading of these manures.
- Livestock numbers and housing type on the farm determine the volume of manure to be stored;
- Area available for land spreading;
- The type of crops grown influences the volume which can be spread and the time of application (e.g. seedbed application, top dressings in spring or during the growing season).

Manure storage has many advantages: Spreading can be restricted to periods when the ground can take wheeled traffic and when crops will accept top dressings. It reduces the risks of surface run-off or pollution into land drains by allowing application to take place when soil conditions are favourable. Manure storing also reduces the frequency of land spreading, providing fewer occasions which may produce a risk of offensive odours. It also minimises the health hazards associated with the spreading of fresh manure, because a storage period of at least 6 weeks reduces any pathogenic micro-organisms (MAFF, 1984). Properly constructed stores reduce the loss of plant nutrients to a minimum. Less NH_3 volatilisation at the storage stage can, however, result in higher losses during landspreading.

There are also some disadvantages associated with manure storage, such as the cost of constructing and maintaining the storage structures. Furthermore, slurry separates into layers during storage (surface crust, middle liquid layer, bottom sludge), which can lead to difficulties when the store is emptied. Rainfall on the large open surface of the slurry and FYM stores increases the eventual volume of waste to be handled. Lastly, large amounts of the N in the manure may be lost due to volatilisation before it is applied to the land.

2.4.6. Landspreading of livestock manures

Disposal of accumulated manures from livestock housing takes place mainly by spreading the slurry or FYM onto either arable fields or grassland. An alternative option for dried poultry manure is burning in power stations/waste incinerators.

An important factor regarding N losses from landspreading of manures is the timing. It has been estimated that over 75% of the total ammoniacal N can be lost following slurry applications to grassland in October/November, through surface run-off on waterlogged soils, volatilisation and leaching (Lauer *et al.*, 1976; MAFF, 1980b; Sommer *et al.*, 1991; Menzi *et al.*, 1998). For applications in December/January, 50% losses have been estimated, and 25% after February/March applications (MAFF, 1980b). The larger autumn and winter losses are due to the plants not utilising the available N until the growing period starts, thus allowing extended periods for volatilisation and leaching.

As most livestock farms have sufficiently large manure storage facilities for at least part of the winter, it is possible to avoid the nutrient losses caused by early applications. Legislation already regulates maximum annual application rates and timing of applications in selected areas (e.g. NSA Scheme: MAFF/DoE, 1990) in Britain. Similar regulations already implemented in some European countries (Sommer *et al.*, 1991; FRAME, 1992). Increasingly, legislation and mandatory as well as voluntary guidelines are provided in the UK to help minimise losses and damage to the environment. Examples for this are the "Codes of good practice" published by the UK Ministry of Agriculture (e.g. MAFF, 1992a and b). From the farmer's point of view, the cost of collection and storage of manures is increasing as new legislation comes into force. These costs will be borne by the farmers whether they fully implement the "Codes of Good Practice" or not. Therefore a central way to offset these additional costs is to reduce the reliance on mineral fertilisers through the efficient use of manures.

The highest losses of NH_3 from manures to the atmosphere occur immediately after the material is spread and during the following day (e.g. Sommer and Olesen, 1991; MAFF, 1992a; Whitehead and Raistrick, 1992; Menzi *et al.*, 1998). Therefore any attempts at reducing the losses have to address the methods of application. Various techniques for more efficient landspreading have been developed: the simplest way to reduce losses is by incorporating the manure into the soil immediately or as soon as possible after spreading.

The most common slurry spreading device is the 'splash plate' mechanism (Figure 2.12a). When directed onto the splash plate, the slurry jet shatters into small drops and releases large quantities of volatile compounds directly into the air. Other techniques require specialised machinery, such as band spreaders or slurry injectors (Figure 2.12b). Band spreaders discharge the slurry at ground level through a series of trailing pipes, while slurry injectors apply slurry in grooves directly into the soil and close the grooves again immediately. Both these methods are very effective, but are restricted to certain conditions, i.e. injecting is not suitable on stony or very heavy soils, or in very dry conditions. Depending on the consistency of the slurry, it may be necessary to remove coarse solids to ensure that the spreaders work properly. Odour and NH_3 emissions can be reduced by 85-90% by using injectors (55-60% by using band spreaders) as compared with a conventional splash plate spreader (MAFF, 1992a).

Another method to reduce NH_3 emissions is through the acidification of slurry prior to spreading (Husted *et al.*, 1991; Stevens *et al.*, 1992). This is due to the solubility equilibrium for gaseous NH_3 (see Section 2.2.). The weather conditions during landspreading and for the first day or so afterwards also have a considerable influence on the amount of N lost through volatilisation, especially when the slurry is not ploughed in (e.g. Jarvis and Pain, 1990; Sommer *et al.*, 1991; Whitehead and Raistrick, 1992; Sommer and Olesen, 1991). This causes higher emission rates during warmer and drier conditions.

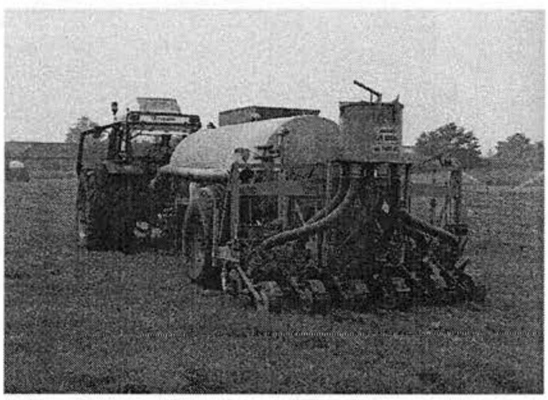


Figure 2.12. (a) Slurry injection (from Frame, 1992)

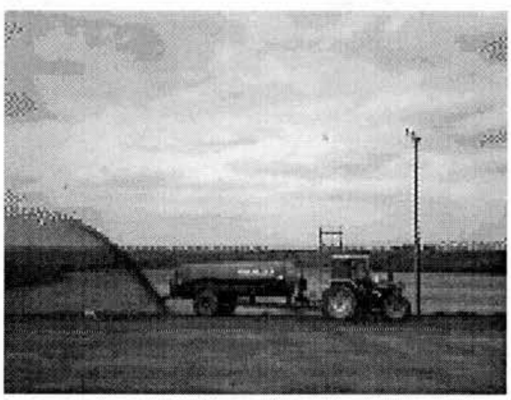


Figure 2.12.(b) Slurry spreading with splash plate spreader

As solid manures cannot be injected in order to minimise NH_3 volatilisation, they should be incorporated into the soil by ploughing as soon as possible after spreading (e.g. MAFF, 1992a; Cowell and ApSimon, 1998). Solid manure is more often applied to arable land or used for reseeding grassland, as incorporation of manure by ploughing-in on established grass swards is not possible without destroying the sward in the process.

2.5. ARABLE FARMING PRACTICES AFFECTING AMMONIA EMISSIONS

Arable farming in the UK is dominated by cereal growing, with wheat and barley as the major crops. Other important crops are oilseed rape, sugar beet, potatoes, crops grown for stock feeding and horticultural crops (GSS, 1996; Figure 2.13.). Stockfeeding or forage crops have played a significant role in the provision of winter feed, but their use has declined throughout the last 50 years and the area for forage crop growing has fallen by 80% (Frame *et al.*, 1995).

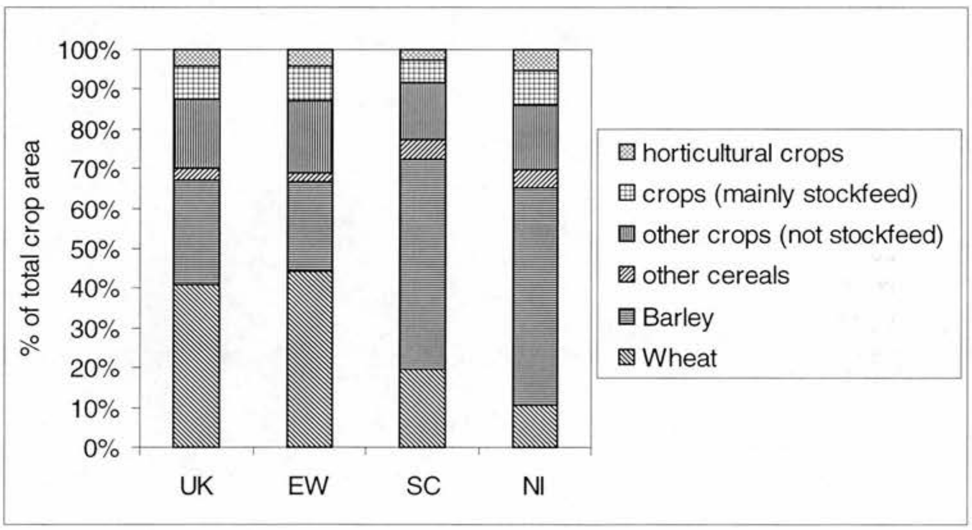


Figure 2.13. Main crops and crop groups grown in the UK, England & Wales, Scotland and Northern Ireland (after GSS, 1996).

Crops require adequate amounts of plant nutrients to ensure optimum yields. Applications of nutrients in excess of the optimum can result in stagnating or even reduced yields. It is therefore important to predict nutrient requirements as accurately as possible. Nutrients may be supplied by commercial fertilisers alone or in combination with organic manures, which are potentially valuable sources of plant food. The application rates of mineral fertiliser should match the crop requirements, taking into account soil nutrient status and any organic manures applied, according to fertiliser recommendations (e.g. MAFF, 1984; MAFF, 1991b; Chadwick, 1998; Nix and Hill, 1997).

The fertiliser rates recommended by ADAS are based on the amounts that will give the highest economic returns. For any particular crop, the rate depends on previous cropping and fertiliser use, soil type, expected yields and the use of organic manures (MAFF, 1987d). Surveys on actual fertiliser practice as opposed to recommended practice are also available. The most important and comprehensive of these for the UK is the annual British Survey of Fertiliser Practice (BSFP, e.g. Burnhill *et al.*, 1996, 1997).

The most important nutrients to encourage plant growth are nitrogen (N), phosphorus (P) and potassium (K). Applications of fertilisers containing N provide a significant source of NH_3 emissions from crops. In most cases, this is the largest N source, if no organic manure is spread onto the field. The magnitude of NH_3 emissions from fertiliser applications depends on the type of fertiliser (i.e. the form in which the N is available) in combination with the N content, the application rate and technique. Furthermore, the climatic conditions and stage of growth of the plants at the time of application and for some time afterwards have a significant influence on the amount of NH_3 volatilised. Nitrogenous fertilisers should be applied at times when the crop can utilise the N, i.e. application should be avoided in the absence of active crop growth during winter, with the exception of some autumn- or winter-sown crops. This helps to minimise not only losses through volatilisation, but also nitrate leaching and run-off.

In general, fertilisers are either of the 'straight' or 'compound' type (MAFF, 1963, 1987d). Straight fertilisers normally contain only one of the main plant nutrients, either nitrogen, phosphorus or potassium. Compound fertilisers are mixtures of two or more nutrients, at various proportions. Compound fertilisers may contain N and P ('NP compounds'), N and K, P and K or all three (NPK). Most compounds are of the NPK type, also known as 'complete fertilisers', implying that they contain the three major nutrients necessary for the crop.

Straight N fertilisers are available as solids or in liquid form. Examples for solid fertilisers are ammonium nitrate (AN), ammonium sulphate (AS) or urea.

Ammonium nitrate is the most commonly used solid fertiliser in Britain. It contains 34 % N, half of which is present in ammonium, half in nitrate form. It absorbs moisture rapidly and is readily available for uptake by the plant roots. When lime is added to help prevent soil acidification, it is referred to as calcium ammonium nitrate (CAN; 21-26% N, 19-20% $CaCO_3$). Ammonium sulphate (21% N) may be slightly less efficient than AN as a top dressing on chalk soils, due to significant losses of N as NH_3 to the air. This is due to direct fertiliser losses generally increasing with higher soil pH, as shown in the combined water solubility and dissociation equilibria (see Equation 2.1. in Section 2.2; Whitehead and Raistrick, 1990; ECETOC, 1994). Urea (46% N) cannot be used directly by crops, and must first be converted to ammonium N as well as to nitrate N in the soil. This provides an opportunity for substantial losses through NH_3 volatilisation (e.g. ECETOC, 1994; Whitehead and Raistrick, 1990; Sutton *et al.*, 1995; Van der Weerden and Jarvis, 1997), though losses from denitrification or nitrate leaching are lower than with AN. In arable situations this effect is more pronounced on chalk and light sandy soils, i.e. with a high soil pH. The NH_3 loss is also higher in warmer and drier conditions. This makes the effectiveness of urea on DM production more variable than that of ammonium nitrate. The use of urea in the UK has increased in recent years, because of its high N concentration and relative cheapness per kg N, compared to ammonium nitrate (Jarvis and Pain, 1990; Frame, 1992). Both AN and urea are widely used as the N constituent of compound NPK fertilisers in the UK.

Liquid N fertilisers are available at various concentrations (MAFF, 1987d). Anhydrous ammonia is the most concentrated form of N fertiliser available with an 82% N content. It is handled as a gas liquefied under high pressure and contained in tankers at pressures of up to 20 bar. Anhydrous ammonia (as other liquid fertilisers) is applied through injection into the soil (usually at about 15 cm depth), where it reverts to its gaseous form. When correctly applied, it is absorbed immediately, and losses to the air are small (ECETOC, 1994; MAFF, 1987d). This type of N fertiliser is less suitable for heavy or stony soils, or soils with a low clay content. The cost of application is greater than for solid fertilisers, because special equipment and training are required for injection (MAFF, 1980c and1987d; Frame, 1992).

Aqueous ammonia (up to 28% N content) is NH_3 dissolved in water under slight pressure. Although it has to be injected into the soil, the much lower pressures involved (compared with anhydrous NH_3) make handling easier and there is less risk of loss during application. It is normally applied in a single dressing in early spring, and the N is then slowly released throughout the summer. Thus additional solid N fertiliser applications for early grass production may be required. Once injected into the soil, it behaves in a similar way to other ammonium containing fertilisers (MAFF, 1987d; Frame, 1992). Other liquid N fertiliser types are liquid urea (up to 20% N) and aqueous N (up to 37 % N) solutions. These are normally as effective as if applied in solid form (MAFF, 1987d).

Fertiliser recommendations for any particular crop not only depend on the crop to be grown, but also on the history of the field in terms of previous crops, soil N status and fertiliser history. Only 1-2% of the soil organic N may become available during the growing season by mineralisation from microbial action, but the amounts range from 20 to 120 kg N ha^{-1}, depending on the soil reserves. Also, considerable amounts of N may be deposited from the atmosphere onto the land by wet or dry deposition (e.g. INDITE, 1994). The largest amounts of wet deposition in Britain were found in the uplands of Wales, northern England and western Scotland (up to 30 kg N ha^{-1} $year^{-1}$). The highest contributions from dry deposition in Britain were recorded in the Midlands and southeast England (10-15 kg N ha^{-1} $year^{-1}$; INDITE, 1994). Furthermore, the symbiotic N-fixation by legumes (see 2.4.2.) as well as free-living soil micro-organisms capable of fixing atmospheric N contribute to the soil N status (approx. 6.7 kg N ha^{-1}; Grigg, 1995). All these sources should be estimated for each field, if optimum N application rates for the next growing season are to be established as precisely as possible (Frame, 1992). Examples of the soil N status, also referred to as Soil N Index (SNI) are given in Table 2.5. (Dampney and Unwin, 1993; MAFF, 1991b), based on previous field history. Usually it is only necessary to consider the last crop grown to determine the field's N status. Only after lucerne, long leys or permanent pasture, which leave longer lasting N residues, is it necessary to take longer histories into account.

Table 2.5. Examples of soil N supply according to previous field history: (long ley = 3 years production or more; ley/arable rotation contains 3 arable crops or less); after Dampney and Unwin (1993) and MAFF (1991b).

(0) Low soil N supply (no organic manures in last 2 years AND)

- four years all arable, last crop a cereal OR
- three years arable in ley/arable rotation OR
- grass, less than 100 kg ha^{-1} N, no clover.

(1) Moderate soil N supply

- four years all arable, last crop a break crop; OR
- reseeded in last 2 years from grass receiving less than 100 kg ha^{-1} N and no clover; OR
- long ley receiving 100-250 kg ha^{-1} N, cut and/or grazed; OR
- long ley receiving over 250 kg ha^{-1} N, cut only; OR
- one year arable or 1-2 year ley receiving less than 250 kg ha^{-1} N, cut and/or grazed in ley/arable rotation

(2) High soil N supply

- one to 2 year or long ley receiving over 250 kg ha^{-1} N, grazed or 1 cut only; OR
- reseeded in last 2 years from long ley receiving over 100 kg ha^{-1} or from sward with high clover content; OR
- any previous cropping with heavy, frequent use of organic manures.

The recommendations for actual mineral fertiliser application rates are guidelines to the amount of N available for crop uptake, which are given by MAFF, ADAS and SAC (e.g. MAFF, 1984; Chadwick, 1998; Nix and Hill, 1997) for each crop, depending on different N indices, expected yield and soil types. The availability of N is variable and is affected by the rate and timing of the application, the weather after spreading and the speed of incorporation into the soil. An example is given in Table 2.6. for winter wheat (a) and spring wheat, spring barley and winter barley (b) (MAFF, 1984).

Table 2.6.(a) Fertiliser recommendations for winter wheat for different target yields with different soil conditions and soil N status (application rates in kg N ha^{-1}); source: MAFF, 1984.

N Index	Soil type	0 (low)	1 (moderate)	2 (high)
Yield up to 7 t ha^{-1}	Sand	175	150	75
	Chalk/limestone	175	150	75
	Deep silty soil	150	50-75	nil
	Clays	150	75	nil
	Other mineral soils	150	100-125	50
	Organic soils	50	nil	nil
	Humose soils	90	45-70	nil
Yield 7-9 t ha^{-1}	Chalk/limestone	225	200	125
	Deep silty soils	200	100-125	nil
	Clays	200	125	nil
	Other mineral soils excl. sandy	200	150-175	100

N Index	Soil type	0 (low)	1 (moderate)	2 (high)
Yield > 9 t ha^{-1}	Chalk/limestone	275	250	175
	Deep silty soils	250	150	50
	Clays	250	175	50
	Other mineral soils excl. sandy	250	200	150

Table 2.6.(b) Fertiliser recommendations for spring wheat, winter barley and spring barley for different soil conditions and soil N status (application rates in kg N ha^{-1}); source: MAFF (1984).

Soil N status	(0) low	(1) moderate	(2) high
Spring wheat			
Sandy/chalk/limestone	150	100	50
Deep silty or clay soils	125	50-75	nil
Other mineral soils	125	75	30
Organic soils	40	nil	nil
Humose soils	70	35	nil
Winter barley			
Sandy/chalk/limestone	160	125	75
Other mineral soils	160	100	40
Organic soils	50	nil	nil
Humose soils	90	45	nil
Spring barley			
Sandy	125	100	50
Chalk/limestone	150	125	50
Other mineral soils	150	100	40
Organic soils	40	nil	nil
Humose soils	70	35	nil

The most important factors for deriving average NH_3 emission source strength estimates from fertiliser N applications are the application rate and the fertiliser type. The recommendations presented above are guidelines only and should not be taken to represent actual farming practice, as only a proportion of farmers adhere to the recommendations. It would be difficult to choose which of the recommended rates would represent actual fertiliser practice as they are quite variable.

The best available source of information on actual fertiliser practice is the British Survey of Fertiliser Practice (BSFP). It provides information on fertiliser use on the main crops and grass grown in mainland Britain. The survey is carried out annually and based upon returns from a sample of approximately 1500 farms. The overall total use of N fertilisers in England and Wales has risen considerably over the last 25 years (Burnhill *et al.*, 1996), both for grassland and arable/tillage. Grassland applications rose from about 75 kg ha^{-1} around 1970 to about 130 kg ha^{-1} during the mid-1980s, with more variability between years, but generally decreasing since the beginning of the 1990s. N fertiliser application rates to crops also rose significantly from about 85 kg ha^{-1} in 1970 to a maximum around the mid-1980s of about 160 kg ha^{-1} N (Burnhill *et al.*, 1996). Since then, the upward trend has flattened out and turned into a slight decline. The annual BSFP reports provide tables of fertiliser application rates for Great Britain as a whole, as well as separate tables for England & Wales and Scotland, for the main crops, crop groups and the main types of grassland. These tables not only contain application rates for nitrogen, but also for phosphates, potassium, organic manures and lime (e.g. Table 2.7.).

A comparison of recommended and actual application rates (Table 2.8.) shows that simply averaging the highest and lowest recommended rates would result in considerable underestimates in most cases, with overestimates occurring only for very few crops. For many crops, there are also considerable variations between England & Wales and Scotland.

Application rates to grass represent an added difficulty, as the grass categories do not match entirely, with the BSFP categories designed to match the grassland categories in the Agricultural Census rather than the categories used in the recommendations. Fertiliser application to grassland also shows the greatest variations, due to large differences in management practices and intended use. Some of the categories distinguished in the BSFP are grazing with mowing, grazing without mowing, silage or hay production with or without grazing (Table 2.9.). The highest N application rates occur on grass swards intended for ensiling without grazing, the lowest for grass which is used for grazing and hay making. Some of the reasons behind this variation have been explained in Section 2.4.2. Furthermore, there are significant variations between the sample fields within each of these categories (see Table 2.10.).

Table 2.7. Fertiliser application rates to crops and grassland in Great Britain in 1996 (kg ha^{-1}). The overall application rate is calculated by the ratio of the total quantity of nutrient used (in kg) to the total extent of the crop area (in ha). These data provide the means to estimate the total tonnage of fertilisers used during the survey year, when combined with total crop area numbers. The average (field) application rate is the rate of nutrient used on fields which received the nutrient. These field specific application rates provide direct evidence of the level and variation in farming practice (Burnhill *et al.*, 1997).

	Crop area receiving dressing (%)				Average field rate (kg/ha)			Overall application rate (kg/ha)			Fields in sample
	N	P_2O_5	K_2O	FYM	N	P_2O_5	K_2O	N	P_2O_5	K_2O	
Spring wheat	100	53	54	28	154	42	49	154	22	27	39
Winter wheat	98	74	70	11	188	67	73	185	49	51	2774
Spring barley	95	77	81	20	100	48	62	95	37	50	444
Winter barley	98	81	82	18	141	63	73	138	51	59	1036
Oats	99	75	73	13	126	69	79	125	51	58	131
Rye	100	63	65	.	126	49	70	126	31	45	26
Early potatoes	96	88	88	25	166	202	235	160	178	207	51
Maincrop potatoes	94	93	95	39	189	196	269	178	183	256	187
Sugar beet	96	60	75	33	112	67	129	107	40	96	337
Oilseed rape	95	77	70	11	200	66	65	190	51	46	596
Linseed	75	31	32	9	71	56	80	53	17	26	96
Forage maize	70	64	51	89	75	72	85	52	46	43	156
Turnips (stock)	88	85	85	40	65	52	70	57	44	59	23
Kale and cow cabbage	97	86	86	58	96	65	53	93	56	45	27
Other roots/green crops	91	87	87	59	88	90	121	80	79	106	33
Peas	10	50	52	10	24	61	71	2	30	37	202
Beans	8	51	44	6	90	65	68	7	33	30	194
Vegetables (brassicae)	92	87	87	10	218	80	201	201	69	176	79
Vegetables (other)	80	74	78	10	119	90	126	96	67	98	65
Small fruit	82	53	78	37	95	85	97	78	45	76	27
Top fruit	76	49	45	.	65	36	50	50	17	23	99
Other tillage	60	51	51	38	106	54	71	63	28	36	45
All tillage	91	73	71	17	163	69	83	148	51	59	6667
Grass under 5 years	96	72	76	52	185	45	73	177	32	56	1049
Grass 5 years and over	85	65	64	51	125	31	41	106	20	26	2979
All grass	87	66	66	51	136	33	47	118	22	31	4028
All crops & grass	89	69	68	35	149	52	65	133	36	45	10695

Table 2.8. Comparison between recommended N application rates and BSFP N application rates for 1996 (in kg N ha^{-1}); after Burnhill *et al.*, 1997 and Nix and Hill, 1997).

	Recommended rates	Mean of range of rates recommended	BSFP 1996 EW (overall rates)	BSFP 1996 SC (overall rates)	BSFP 1996 GB (overall rates)
Spring wheat	30-200	115	141	..	140
Winter wheat	50-275	162.5	192	208	193
Spring barley	40-175	107.5	100	97	99
Winter barley	40-200	120	141	178	144
Potatoes - early	80-200	140	170	..	171
Potatoes - maincrop	50-250	150	183	151	176
Sugar beet	25-130	77.5	118	..	118
Oilseed rape	50-275	162.5	187	190	188
Kale & cow cabbage	75-125	100	93	..	104
Turnips (stockfeed)	50-150	100	85	82	83
Ley grass (> 1year)	50-300	175	191	143	175
					(grass < 5 yr.)
Permanent grass	50-300	175	105	100	104
					(grass ≥ 5 yr.)

Table 2.9. Average fertiliser practice by grassland utilisation for England & Wales and Scotland in 1996 (from: Burnhill *et al.*, 1997).

Grassland type	Overall N application rate (kg ha^{-1}) England and Wales 1996	Overall N application rate (kg ha^{-1}) Scotland 1996
Grazed-not mown	93	77
Grazed-mown	148	148
All grazings	114	92
Cut for seed-grazed	..	..
Cut for seed-not grazed	145	..
All cut for seed	145	..
Cut for silage-grazed	175	164
Cut for silage-not grazed	170	159
All cut for silage	174	162
Cut for hay-grazed	91	112
Cut for hay-not grazed	91	131
All cut for hay	91	118
All mowings	149	150
All grass	118	100

Table 2.10. Percentage of crop area by field N application rate for England and Wales in 1996 (from Burnhill *et al.*, 1997).

					kg/ha							
row %	0	<25	25-	50-	75-	100-	125-	150-	200-	250-	300-	400+	Fields in sample
Spring wheat	.	.	8	1	20	2	11	36	4	18	.	.	39
Winter wheat	2	.	1	2	3	4	7	39	36	4	2	1	2774
Spring barley	5	.	6	12	26	29	13	6	.	1	.	.	444
Winter barley	2	.	3	3	8	16	27	33	6	2	.	.	1036
Oats	1	.	3	4	16	19	38	14	.	4	.	1	131
Rye	.	.	.	2	41	23	14	11	.	10	.	.	26
Early potatoes	4	.	12	.	4	.	.	48	27	2	3	.	51
Maincrop potatoes	6	1	3	1	4	5	2	34	33	6	3	.	187
Sugar beet	4	.	12	9	17	18	27	8	2	1	2	.	337
Oilseed rape	5	.	1	3	3	2	8	24	36	15	2	.	596
Linseed	25	2	30	18	11	2	4	7	.	.	.	.	96
Forage maize	30	12	10	16	9	6	14	3	.	.	.	.	156
Turnips (stock)	12	6	14	36	28	4	.	.	.	.	.	.	23
Kale and cow cabbage	3	.	3	23	21	16	31	1	2	.	.	.	27
Other roots/green crops	9	.	5	36	17	11	14	8	.	.	.	.	33
Peas	90	7	3	.	.	.	.	.	.	.	.	.	202
Beans	92	2	2	1	.	.	1	2	.	.	.	.	194
Vegetables (brassicae)	8	.	1	8	3	2	14	7	22	29	.	5	79
Vegetables (other)	20	6	4	9	11	17	4	27	3	1	.	.	65
Small fruit	18	24	3	1	15	7	3	29	.	.	.	.	27
Top fruit	24	35	4	5	8	3	16	3	1	.	.	.	99
Other tillage	40	.	12	18	6	5	1	11	8	.	.	.	45
All tillage	9	1	3	4	7	8	12	28	22	4	1	.	6667
Grass under 5 yrs	4	.	5	9	8	9	13	13	14	12	11	2	1049
Grass 5 yrs and over	15	1	11	18	12	8	8	11	7	3	4	1	2979
All grass	13	1	9	16	12	8	9	11	8	4	5	2	4028
All crops & grass	11	1	7	11	9	8	10	20	15	4	3	1	10695

2.6. SUMMARY AND CONCLUSIONS

Agriculture in the United Kingdom is very diverse, compared with some other European countries such as the Netherlands or Denmark. The major influences contributing to this diversity are environmental factors such as topography, climate (especially temperature and precipitation) and soil conditions, which are themselves very variable across the country. Within the limitations imposed by local environmental conditions, agriculture and agricultural practice have developed over many centuries, with significant changes taking place during the last century (intensification, mechanisation and scientific advances). Linked to this diversity in farming is a variability in NH_3 emissions. Whether they specialise in arable crops, horticulture, intensive dairying, pigs, poultry or extensive hill sheep farming, all farms contribute to NH_3 emissions.

The type of husbandry and the breed both have a significant impact on NH_3 emissions for any particular livestock category. Important parts of livestock husbandry are livestock grazing and all the related aspects of grassland management, livestock housing, the storage of livestock manures and manure spreading. The major influences on NH_3 emission source strength from crops and grassland are fertiliser N application rates and the types of N fertiliser used. The method and timing of applications also play important roles.

This chapter provides a general overview of the main processes governing NH_3 emissions, and links these with the environmental factors and components of farming practice that influence agricultural diversity and consequently NH_3 emissions in the UK. In so doing, it has shown the degree of variability and uncertainty associated with any attempt to quantify the magnitude and distribution of NH_3 emitted as a result of agricultural activity. This provides the basis for discussing and assessing the wide range of emission source strength data provided in the literature (see Chapter 3). Furthermore, knowledge about the state of UK agriculture is helpful when investigating emission abatement potential (Chapter 7) and for examining uncertainties in the emission inventory (Chapters 9 and 10).

Chapter 3

Estimates of ammonia emission source strength in the UK

3.1. INTRODUCTION

Quantifying the magnitude of NH_3 emissions and the underlying major uncertainties is one of the main aims of ammonia research, which is necessary to develop budgets and atmospheric transport models as well as formulate effective abatement measures. Despite considerable research efforts, especially over the last few decades, large uncertainties still remain, not only regarding the total magnitude of NH_3 emissions, but also concerning the contributions by the major sources. In this chapter, ammonia source strength estimates and related uncertainties in the relevant literature are critically reviewed, to highlight the main similarities and differences. The assumptions underlying NH_3 source strength estimates in various recent studies are also compared with knowledge of agricultural practice in the UK (see Chapter 2), to check their applicability to UK circumstances, as well as to emphasise the major uncertainties.

The main sources of NH_3 emissions in the UK originate in agriculture, chiefly in livestock husbandry, with smaller contributions from fertiliser applications to crops and grassland and non-agricultural sources, as described earlier (see Chapter 1). UK NH_3 emissions have been estimated between 180 kt NH_3-N year^{-1} (Jarvis and Pain, 1990; from agricultural sources only) and 440 kt NH_3-N year^{-1} (Eggleston, 1992; including non-agricultural sources) by recent studies (see INDITE, 1994; Sutton *et al.*, 1995; RGAR, 1997; summarised in Table 1.1., Section 1.4.). Earlier studies by Healy *et al.* (1970), Hood (1982), Fisher (1984), Ryden *et al.* (1987) and Buijsman *et al.* (1987) are now considered out of date (Sutton *et al.*, 1995). The uncertainty regarding the magnitude of NH_3 emissions for the UK in these earlier studies was even larger, ranging from 70-105 kt NH_3-N (Healy *et al.*, 1970) 595 kt NH_3-N (Hood, 1982).

The contributions of the main UK ammonia sources as estimated in recent inventories (ECETOC, 1994; Sutton *et al.*, 1995; DoE, 1995; TFEI, 1996; BBSRC, 1997a; BBSRC, 1997b) are shown in Table 3.1. and Figures 3.1. and 3.2. The emission source strength estimates per animal derived from these studies were applied to livestock numbers from 1996 (taken from BBSRC, 1997b). Ammonia emissions from fertiliser application to crops and cut grassland were estimated on the basis of UK fertiliser consumption figures for the main fertiliser types in 1996, taken from BBSRC (1997b). Estimates for non-agricultural emissions, where provided by the authors of the inventories presented in Table 3.1., were not converted to 1996 estimates.

Table 3.1. Ammonia emissions in the UK (in kt NH_3-N $year^{-1}$); source strength estimates from ECETOC (1994), Sutton *et al.* (1995), DoE (1995), TFEI (1996), BBSRC (1997a) and BBSRC (1997b) applied to livestock numbers and fertiliser use for the base year 1996; non-agricultural emission estimates by different authors not corrected to base year 1996.

Main source types	Livestock nos. (1996)	ECETOC (1994)	Sutton *et al.*[a] (1995)	DoE (1995)	TFEI (1996)	BBSRC (1997a)	BBSRC (1997b)
Base year of original study	-	1990	1988	1993	-	1993	1996
Cattle	11,953,300	273.7	203.2 (99-323)	134.2	177.9	106.9	125.8
Sheep	41,611,200	32.9	34.3 (13-50)	15.8	22.9	12.0	12.9
Pigs	7,496,100	25.9	32.2 (23-43)	23.8	29.9	21.5	25.7
Poultry	141,598,000	21.9	37.2 (21-43)	26.9	42.8	30.1	44.2
Total livestock	-	354.5	306.9 (156-455)	200.8	273.4	170.5	208.6
Fertilisers	-	50.7	20.9 (15-51)	31.9	24.7	17.7	17.7
Total agriculture	-	405.2	327.8 (171-506)	232.7	298.1	188.2	226.3
Non-agricultural sources	-	40.5	36.2 (13-70)	32.9	-	-	-
Total	-	445.2	364.0 (181-576)	265.6	298.1	188.2	226.3

[a] uncertainty estimates in brackets

The studies included in Table 3.1. and Figures 3.1. and 3.2. as well as most other inventories mentioned earlier agree that the largest contribution to the total agricultural livestock emissions is from cattle (~60-75%). Poultry, pigs and sheep together provide the remaining ~25-40% (see Figure 3.2.). In absolute terms, however, the different inventories show large discrepancies, as can be seen in Figure 3.1. Emissions from the application of fertilisers to crops and cut grassland are estimated to contribute 6-14% of the total agricultural emissions. All agricultural ammonia sources together are estimated to provide 88-91% of the total emissions by the studies quoted in Table 3.1.

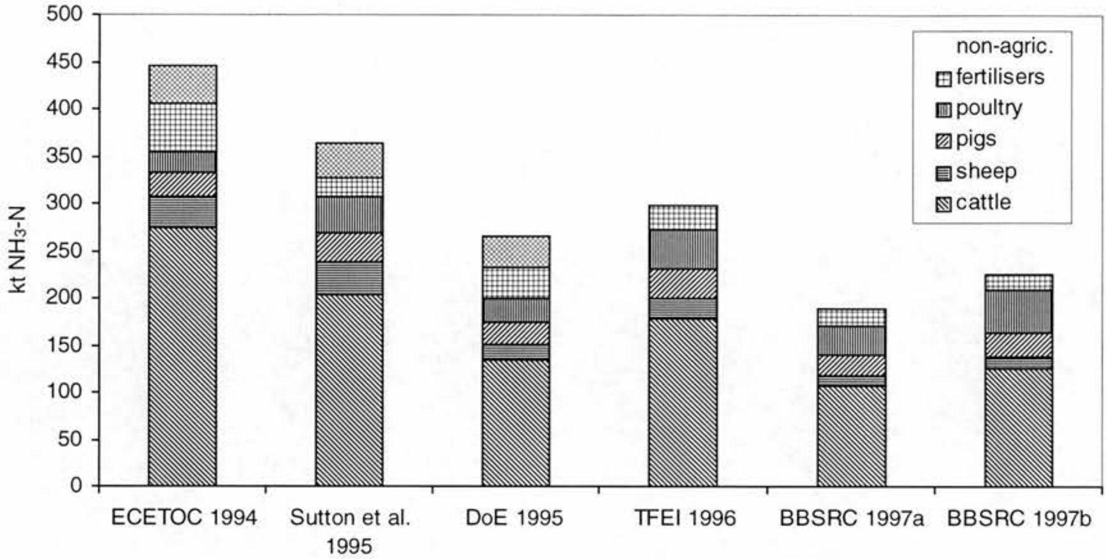


Figure 3.1. Contributions from the main source types to UK NH_3-N emissions according to different inventories' source strength data, projected for 1996 animal numbers and fertiliser use; non-agricultural emissions not corrected for the base year 1996; (derived from Table 3.1 above).

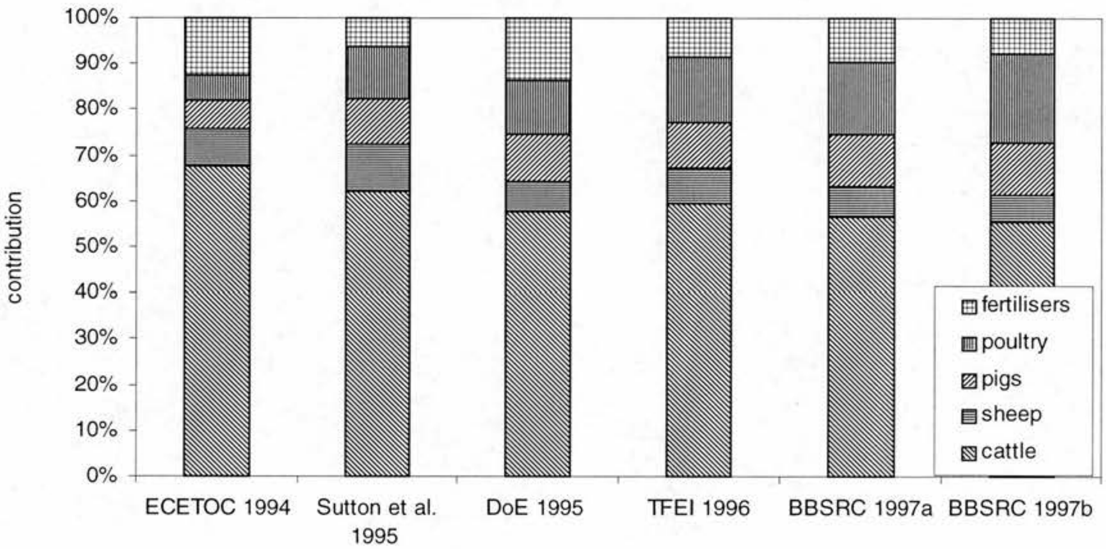


Figure 3.2. Contributions from the main source types to UK agricultural NH_3-N emissions according to different inventories source strength data, projected for 1996 animal numbers and fertiliser use, as % of total emissions from these source types; (derived from Table 3.1 above).

In the following paragraphs, the main NH_3 emission inventories considered here are introduced briefly. Subsequently, the underlying factors contributing to the differences between their estimates are examined in more detail, separately for the main livestock types (cattle, sheep, pigs and poultry), fertiliser emissions from crops and grassland and non-agricultural sources.

3.1.1. Earlier inventories

The NH_3 emission inventories by Jarvis and Pain (1990) and Asman (1992b) are the main earlier studies discussed here. Jarvis and Pain (1990) provided a UK inventory that was partly based on British experimental studies, rather than Dutch NH_3 estimates as largely used by Asman (1992b). This resulted in the lowest of the earlier UK NH_3 emission inventories at 186 kt NH_3-N from agricultural sources (Sutton *et al.*, 1995). Jarvis and Pain (1990) provided the predecessor study for the BBSRC inventories discussed in Section 3.1.6. (BBSRC 1997a, b). Using Dutch source strength estimates, which are based on more intensive agricultural practice, Asman (1992b) estimated UK agricultural NH_3 emissions at 385 kt NH_3-N.

3.1.2. ECETOC 1994

This report on European NH_3 emissions was produced by the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC) in Brussels. The main aim was to update European NH_3 emission estimates for all Western European countries separately as well as for Western Europe as a whole and hence provide a more detailed picture. For the UK, this inventory provides the largest estimates of all the studies outlined above (445 kt NH_3-N), mainly due to a large estimate of cattle NH_3 emissions.

3.1.3. Sutton et al. 1995

Sutton *et al.* (1995) developed a set of best estimates from critically reviewing a collection of NH_3 emission inventories, mainly from the UK, but also considering Dutch and general/averaged European inventories. They also assigned uncertainty margins to their best estimates, depending on the level of agreement between the different studies and a critical assessment of the underlying input estimates. Thus they also provided best and worst case scenarios.

3.1.4. DoE 1995

This inventory was derived from an agreement between the Ministry of Agriculture, Fisheries and Food and the Department of the Environment (DoE, 1995), for submission to European bodies (EMEP) as the "official" UK NH_3 emissions (see Table 3.2.). The figures contained in this agreement are rounded averages of several British studies (Institute for Grassland and Environmental Research North Wyke (IGER/BBSRC) Imperial College London (IC), Institute of Terrestrial Ecology (ITE) and the Agricultural Development and Advice Service (ADAS)). No details are given as to how the main emission sources (cattle, sheep, pigs, poultry, fertiliser and non-agricultural) were derived and scientifically backed up. Thus, no source strength estimates for individual sources, i.e. one sheep or pig, were provided by this inventory. These were derived from the averaged totals per source class (see Table 3.2.; Dragosits *et al.*, 1996b). Recently these 'official' emission source strength estimates were updated and scientifically underpinned (see BBSRC 1997b, Section 3.1.6. below).

Table 3.2. Total NH_3 emissions in the UK (base year 1993) according to DoE, 1995 and equivalent emission factors derived (NB: Emissions are given in units of NH_3, not NH_3-N. The latter can be obtained by using the conversion factor c = 14/17)

Source category	Livestock numbers (thousands)	Total emissions (kt NH_3 $year^{-1}$)	Emission per animal (kg NH_3 $animal^{-1}$ $year^{-1}$)
Cattle & calves	11,729.0	160	13.6
Pigs	7,753.8	30	3.87
Sheep & lambs	43,901.0	20	0.46
Fowls	130,043.0	30	0.23
Tillage & cut grass	-	40	-
Non-agricultural sources	-	40	-
Total emissions	-	320	-

For the main part of this thesis, i.e. the modelling of the spatial distribution of NH_3 emissions for the UK, only one set of emission source strength estimates was used. The reason for this was to keep the main issues under investigation separate as (a) the spatial distribution of emissions (Chapters 4-7) and (b) the uncertainties involved in spatial NH_3 emission inventories (Chapters 9 and 10). For this purpose, the 'official' estimates (DoE, 1995) were chosen for the spatial emissions model, due to the fact that they were a compromise agreed between the main UK research groups involved in ammonia research. Their main advantage is also their main disadvantage, i.e. the fact that they were compromised estimates, and that they are not transparently underpinned with figures from research. However, it was felt that this disadvantage did not outweigh the advantages described above. Throughout the work described in this thesis, great care was taken that the uncertainties of any particular set of emission source strength estimates are not forgotten. These uncertainties are further investigated in the following sections for the main emission sources. Furthermore, Chapters 9 and 10 are devoted entirely to the subject of uncertainties found in all aspects of the spatial distribution of NH_3 emissions. This also includes the uncertainties encountered by applying different sets of emission source strength data within the spatial model developed in subsequent chapters.

3.1.5. TFEI 1996

The figures derived under the heading TFEI for Table 3.1. and Figures 3.1. and 3.2. above originate from the 'simple methodology' guidelines for calculating NH_3 emission inventories, as outlined in the 'Atmospheric Emissions Inventory Guide Book' (TFEI, 1996). These guidelines were developed by expert panels for a large number of different pollutants, as part of this joint EMEP/CORINAIR publication. Separate chapters are dedicated to methods for calculating NH_3 emissions from manure management, cultures with and without fertilisers and appendices containing tabulated emission source strength data, with options for simpler and more complex methods.

A new version of this publication is in preparation; however, the emission source strength estimates regarding livestock for the simple methodology appear to remain unchanged. For a more complex and accurate methodology, a new detailed spreadsheet model has been developed (Cowell, 1998) which will be included in the new edition of the Guidebook.

3.1.6. BBSRC 1997a and 1997b

The inventory referred to here as BBSRC (1997a) was derived from a detailed study at the Institute of Grassland and Environmental Research (IGER), North Wyke (Pain *et al.,* 1998). It is based on earlier work on NH_3 emission inventories for UK agricultural sources at IGER (e.g. Jarvis & Pain, 1990). The authors claim one of its main advantages is that the emission source strength estimates used in the study were derived almost entirely from field experiments, with mainly British contributions where possible to make it more relevant. The inventory was constructed using a large spreadsheet for the base year of 1993, detailing every input to the model and its origin, which is extremely useful for comparing the results with other studies and for helping to explain the reasons behind any differences occurring. The spreadsheet was updated frequently, and the version discussed here as BBSRC (1997a) was developed in April 1997. This inventory provides the lowest UK estimate of total NH_3 emissions from the main livestock sources of all the studies outlined above.

The inventory referred to here as BBSRC (1997b) is an updated version of BBSRC (1997a). The base year was updated from 1993 to 1996, and the version discussed here was developed in December 1997.

3.2. AMMONIA EMISSION ESTIMATES FOR LIVESTOCK HUSBANDRY

3.2.1. Ammonia emission estimates for cattle

Cattle provide the largest single contribution to the total NH_3 emissions in the UK (Table 3.1. and Figures 3.1. and 3.2.). Estimates derived from some of the latest studies range from 107 to 274 kt NH_3-N for 1996, according to Table 3.1. (above), the lowest estimate derived from BBSRC (1997a), the highest from ECETOC (1994). This large range in the total magnitude of emissions from cattle also provides the largest uncertainty. Actual uncertainties may be larger than the range of values given by the different studies. Sutton *et al.* (1995), for instance, estimated an uncertainty range of 98-320 kt NH_3-N for their best estimate of 202 kt NH_3-N, based on critical assessment of the underlying input parameters. It is therefore extremely important to investigate the reasons behind the differences between the studies shown in Table 3.1.

Ammonia emissions from livestock in general have been calculated using different methods. In the early inventories, simply total emissions were given (e.g. Healy *et al.*, 1970). Later emission inventories (e.g. Jarvis and Pain, 1990; Klaassen, 1992) usually calculated emission source strength by estimating losses for each of the stages of emission in average husbandry conditions and adding these individual losses together to achieve a total emission "factor" per animal (Sutton *et al.*, 1995). The losses during livestock housing, manure storage, manure spreading and grazing may be expressed as component emission factors, which are very useful for examining the differences between studies. More recently, Asman (1992b) and Sutton *et al.* (1995), for instance, started analysing the flow of N through the livestock husbandry system following excretion. Thus the contributions of the component emission factors are linked to the N available for volatilisation at each stage. This approach has been used by TFEI (1996) and Cowell (1998).

By comparing N excretion estimates and subsequently following the flow of N after excretion through the different stages of manure management with associated loss rates at each stage, a comparison between the different studies is possible (Table 3.3.). Some of the authors have not explicitly provided all the information necessary for this analysis, however, most of it is implicitly contained in their calculations and can be inferred. The parameters listed in Table 3.3. can be visualised for better understanding as illustrated in Figure 3.3.

Another difference between the studies discussed here is the level of detail regarding livestock types and classes. Some inventories such as the BBSRC studies provide data and calculations for up to 10 different cattle subclasses, whereas others distinguish mainly between 2 classes, i.e. dairy cows and other cattle, or give estimates for 'average cattle'.

Some of the main differences in emissions from different cattle subclasses are explained by the total amount of N excreted, which in turn is mainly dependent on the breed, age, size and N content in the animal feed. Other reasons for differences in estimates of emission source strength may be found in the conditions and durations assumed for livestock housing and manure storage (e.g. slurry or FYM systems), or the equipment and techniques used for landspreading of manures (see also Chapter 2). Further influences are the environmental conditions affecting volatilisation of NH_3 excreted on the pastures or spread onto crops and grassland spreading (such as temperature, wind conditions, soil parameters).

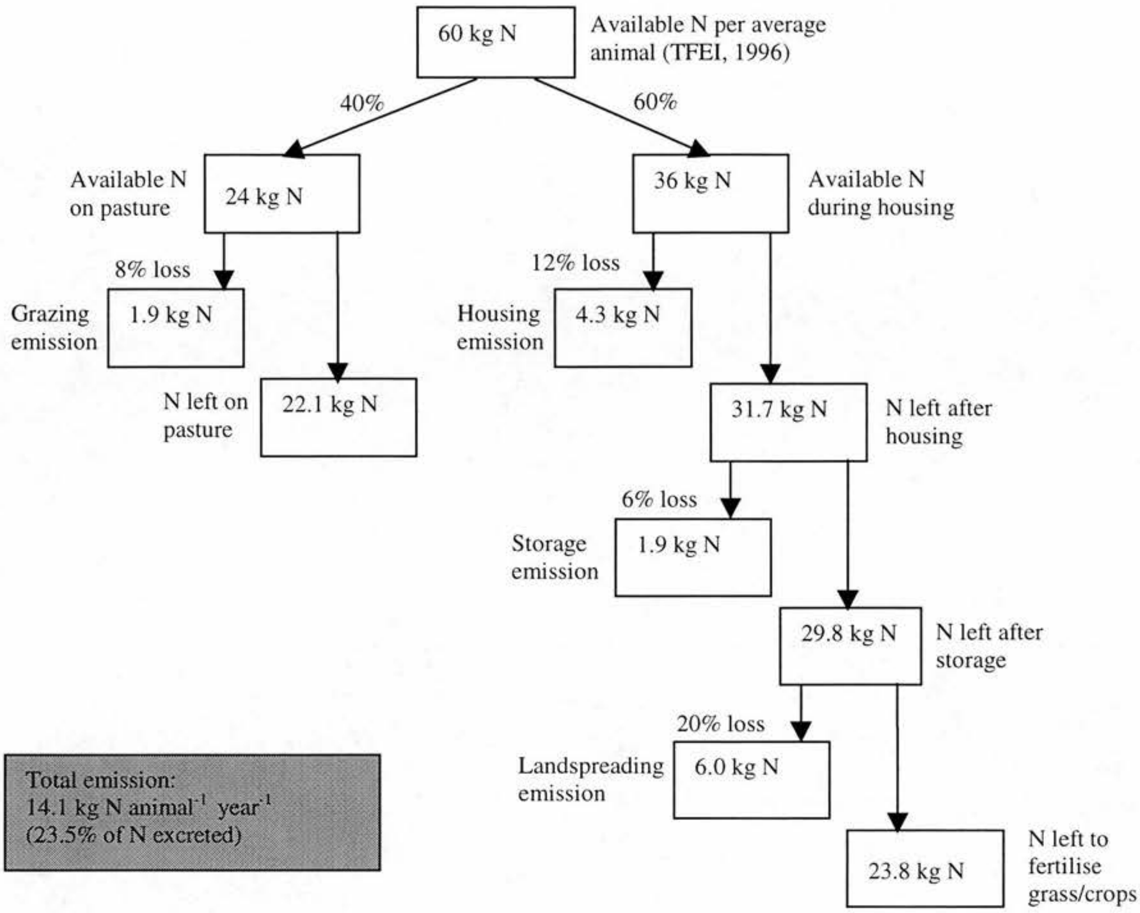


Figure 3.3. The flow of N for estimating NH_3 emissions from average cattle, derived from data of TFEI (1996).

Table 3.3. Ammonia emissions from cattle according to different studies showing the N flow: N excretion rates, absolute and relative component losses (units: *italic* font: % volatilisation rate; **bold** font: emissions in kg NH_3-N animal^{-1} year^{-1}; normal font: available N in kg NH_3-N animal^{-1} year^{-1}).

Study	BBSRC	BBSRC	Asman	ECETOC	TFEI	BBSRC	BBSRC	TFEI	BBSRC	BBSRC	Sutton *et al.*	ECETOC	Jarvis & Pain	Buijsman *et al.*	TFEI
Year of publication	1997b	1997a	1992b	1994	1996	1997b	1997a	1996	1997b	1997a	1995	1994	1990	1987	1996
Cattle type	Dairy cows	Dairy cows	Dairy cows	Dairy cows	Dairy cows	Beef cattle	Beef cattle	Other cattle	Avg. cattle	Avg. cattle	Avg. cattle	Avg. cattle	Avg. cattle	Avg. cattle	Avg. cattle
Total N excreted	97.2	107.2	139	122	100	26.8	25.9	50	62.1	62.8	65	80.7	43.8	48	60
Field - % avail. N	50%	50%	42%	44%	40%	54%	56%	40%	50%	50%	50%	41%	50%	38%	40%
Field - available N	48.6	53.6	58.9	54.2	40	14.4	14.4	20	31.1	31.4	32.5	33	21.9	18	24
Field loss rate	*4.9%*	*1.9%*	*8.0%*	*8.0%*	*8.0%*	*2.8%*	*0.7%*	*8.0%*	*3.9%*	*1.3%*	*8.9%*	*9.7%*	*10.5%*	*24.4%*	*8.0%*
Grazing emission	**2.4**	**1**	**4.7**	**4.3**	**3.2**	**0.4**	**0.1**	**1.6**	**1.2**	**0.4**	**2.9**	**3.2**	**2.3**	**4.4**	**1.9**
Housing - avail. N	48.6	53.6	80.1	67.8	60	12.4	11.5	30	31.1	31.4	32.5	47.7	21.9	30	36
Housing loss rate	*14.4%*	*9.3%*	*13.2%*	*18.3%*	*12.0%*	*23.4%*	*12.2%*	*12.0%*	*11.0%*	*7.0%*	*15.1%*	*7.8%*	*13.7%*	*10.0%*	*12.0%*
Housing emission	**7**	**5**	**10.6**	**12.4**	**7.2**	**2.9**	**1.4**	**3.6**	**3.4**	**2.2**	**4.9**	**3.7**	**3**	**3**	**4.3**
N avail. before storage	41.6	48.6	69.5	55.4	52.8	9.5	10.1	26.4	27.7	29.2	27.6	44	18.9	27	31.7
Storage loss rate	*7.9%*	*6.8%*	*0.0%*	*0.0%*	*6.0%*	*7.4%*	*6.9%*	*6.0%*	*5.1%*	*5.1%*	*10.0%*	*10.9%*	*0.0%*	*0.0%*	*6.0%*
Storage emission	**3.3**	**3.3**	**0**	**0**	**3.2**	**0.7**	**0.7**	**1.6**	**1.4**	**1.5**	**2.8**	**4.8**	**0**	**0**	**1.9**
N avail. before spreading	38.3	45.3	69.5	55.4	49.6	8.8	9.4	24.8	26.3	27.7	24.9	39.2	18.9	27	29.8
Spreading loss rate	*23.0%*	*21.2%*	*25.0%*	*28.5%*	*20.0%*	*13.8%*	*13.8%*	*20.0%*	*14.9%*	*15.0%*	*25.2%*	*28.6%*	*18.0%*	*14.4%*	*20.0%*
Spreading emission	**8.8**	**9.6**	**17.4**	**15.8**	**9.9**	**1.2**	**1.4**	**5.0**	**3.9**	**4.2**	**6.3**	**11.2**	**3.4**	**3.9**	**6.0**
N excreted & not volatilised	75.7	88.3	106.3	89.5	76.5	21.6	22.3	38.3	52.2	54.6	48.2	57.8	35.1	36.7	45.9
Total emission	**21.5**	**18.9**	**32.7**	**32.5**	**23.5**	**5.2**	**3.6**	**11.7**	**9.9**	**8.3**	**16.8**	**22.9**	**8.7**	**11.3**	**14.1**
% of total N excreted volatilised	*22.1%*	*17.6%*	*23.5%*	*26.7%*	*23.5%*	*19.4%*	*13.9%*	*23.5%*	*15.9%*	*13.1%*	*25.8%*	*28.4%*	*19.9%*	*23.5%*	*23.5%*

Some of the studies are clearly related to one another, such as the 2 versions of the BBSRC spreadsheets (1997a, b) as well as Jarvis and Pain (1990), which explains some of the similarities. Also, Asman (1992b), Sutton *et al.* (1995) and ECETOC (1994) are quoted as major references for the CORINAIR/EMEP study (TFEI, 1996) at the higher end of the estimates shown above. For a more in-depth investigation of the differences between the studies, dairy cows, non-dairy cows (i.e. all other cattle) and average cattle were considered separately (see also Figures 3.4. and 3.5.).

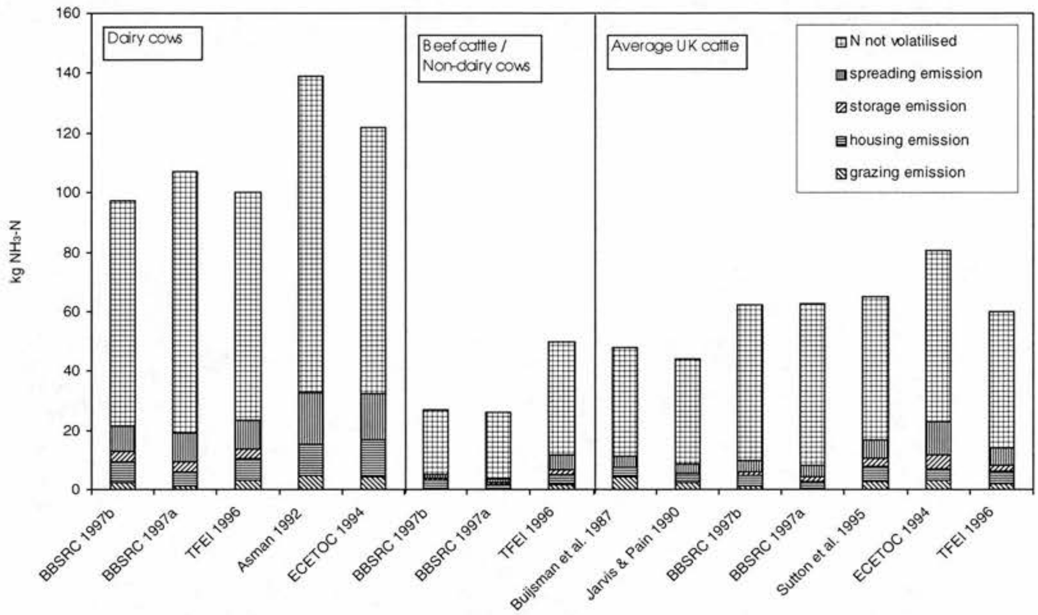


Figure 3.4. Component NH_3 losses from cattle (grazing, housing, manure storage and landspreading of manures) in kg NH_3-N animal^{-1} year^{-1} (derived from Table 3.3. above).

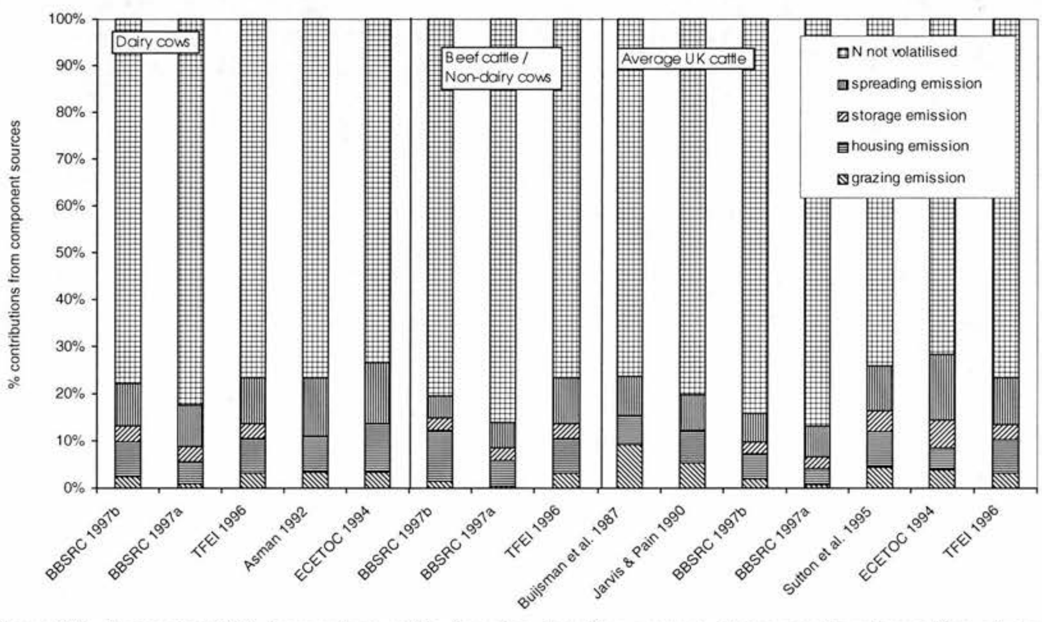


Figure 3.5. Component NH_3 losses from cattle (grazing, housing, manure storage and landspreading of manures) as % of N excreted (derived from Table 3.3. above).

a) Dairy cows

The N excretion rates for dairy cows in Table 3.3. above range from 97.2 kg N animal^{-1} year^{-1} (BBSRC, 1997b) to 139 kg (Asman, 1992b), with earlier BBSRC estimates of 107 kg (BBSRC, 1997a), TFEI (1996) estimates of 100 kg and ECETOC (1994) assuming a rate of 122 kg (see Sutton *et al.* (1995) for other estimates). The N excretion rate is visualised in the total height of the columns in Figure 3.4. With N excretion rates providing the basis for the calculation of NH_3 losses, i.e. the pool of N available for volatilisation, the substantial differences in N excreted between the studies can account for a significant difference in subsequent losses of NH_3.

Considering the absolute total emissions from dairy cows, the range is equally large, with estimates from 18.9 kg N (BBSRC, 1997a) to 32.7 kg N animal^{-1} year^{-1} (Asman, 1992b). However, the estimated relative amounts of N emitted, i.e. the proportions of excreted N lost, are much more similar: the lowest values are again provided by BBSRC (1997a), who estimated the total losses per cow at 17.6%. The highest values are quoted by ECETOC (1994) at 26.7%, with the other studies (Asman, 1992b; TFEI, 1996; BBSRC, 1997b) close together at 23.5% to 24.6% respectively. If it is assumed that the low values estimated by the earlier BBSRC study (1997a) are superseded by the newer version, then the range is very small indeed at 23.5% to 26.7%.

This suggests that the main overall disagreement between the studies is in the total amount of N excreted by dairy cows. Webb (1995) criticised the N excretion rate (122 kg N) proposed by ECETOC (1994) as too high for UK average conditions, as it assumed a fertiliser input of 250 kg N ha^{-1}. This rate was based on monitoring above-average dairy herds (Jarvis, 1993). Webb (1995) suggested a more suitable N excretion rate of around 108 kg N per cow, based on ADAS estimates of typical dietary intakes and average milk yield and an average N fertiliser input of ~175 kg N ha^{-1} (from the BSFP) to grassland. This is consistent with the two BBSRC (1997a, 1997b) and the TFEI (1996) estimates, which are either slightly above or below this figure, and to which Webb contributed. Asman (1992b) and ECETOC (1994) may be overestimating the amount of N available for emission from dairy cow excretion rates (139 kg and 122 kg N per cow respectively), and BBSRC (1997b) may be underestimating at 97 kg N per cow. This uncertainty regarding N excretion rates suggests that more research is needed to improve the emission estimates derived from them.

Of the three main component emissions discussed here (grazing, housing & storage, landspreading), grazing loss rates from livestock are the smallest. The more time animals spend outdoors, the smaller their overall emission rates are estimated to be. As discussed in Chapter 2, the period cattle spend grazing outdoors may vary greatly between different areas of the country, depending on environmental conditions and farming practice. The studies quoted above assume the average proportion of the year spent outdoors for dairy cows at 40-44%. This partly takes milking time indoors during the grazing period into account, which is estimated to contribute 20% (=10 kg) of the total excreted N during the grazing period (TFEI, 1996). The BBSRC studies account for this separately in their spreadsheets under "yard emissions". However, it may be argued that the indoor milking of dairy cows during summer is only partly accounted for by these estimates, which give this component a constant fraction of the annual emissions with the same volatilisation rate. It has been shown, however, that higher temperatures during summer result in higher emission rates from the N excreted indoors (see Section 2.2.). Furthermore, although only a relatively small amount of the total annually excreted N is excreted indoors during summer milking of dairy cows, the dirty surfaces continue to emit after the cows have gone outside again.

Of the N available for NH_3 volatilisation on grazed pastures, the BBSRC studies (1997a, 1997b) again provide the lowest estimates, with 2.1% and 5.6% respectively. It is assumed that the earlier very low estimate has been superseded by the later one. The other main studies estimate grazing losses at 8% of the available N (Asman, 1992b; ECETOC, 1994; TFEI, 1996). The range of 5.6-8% may be explained by the different N fertiliser input rates to grassland assumed in the different studies. As stated in Chapter 2, emissions from grazed pastures are proportional to the N fertiliser input, due to a) higher loss rates at a higher fertiliser input level from the pasture itself, b) higher loss rates per animal at a higher N level in the feed and c) higher stocking rates at higher N input levels and therefore more emission potential.

Jarvis and Bussink (1990) first proposed a function for emission estimates from grazed grassland (Equation 3.1.). This function was updated recently by Pain *et al.* (1997) and estimates both emissions from fertiliser N applied to the grassland and subsequent emissions from excreta deposited by livestock during grazing (see also Figure 3.6.):

$$E = -34.3 + 31.3\,(1.00153)^{N} \qquad [3.1.]$$

where E = NH_3-N emission (in kg ha^{-1})
N = N fertiliser input (in kg N ha^{-1})

This equation was derived almost entirely from field measurements of NH_3 emissions in north-western Europe. The authors state that it should only be used where fertiliser N applications are > 60 kg ha^{-1} $year^{-1}$. Where annual applications of fertiliser N are less than this, emissions from grazed swards are estimated as zero or imply net deposition. The equation does not distinguish between emissions from grazing cattle and sheep. It also does not give emission source strength estimates per animal, but does show how the emission rate increases with increased fertiliser N input.

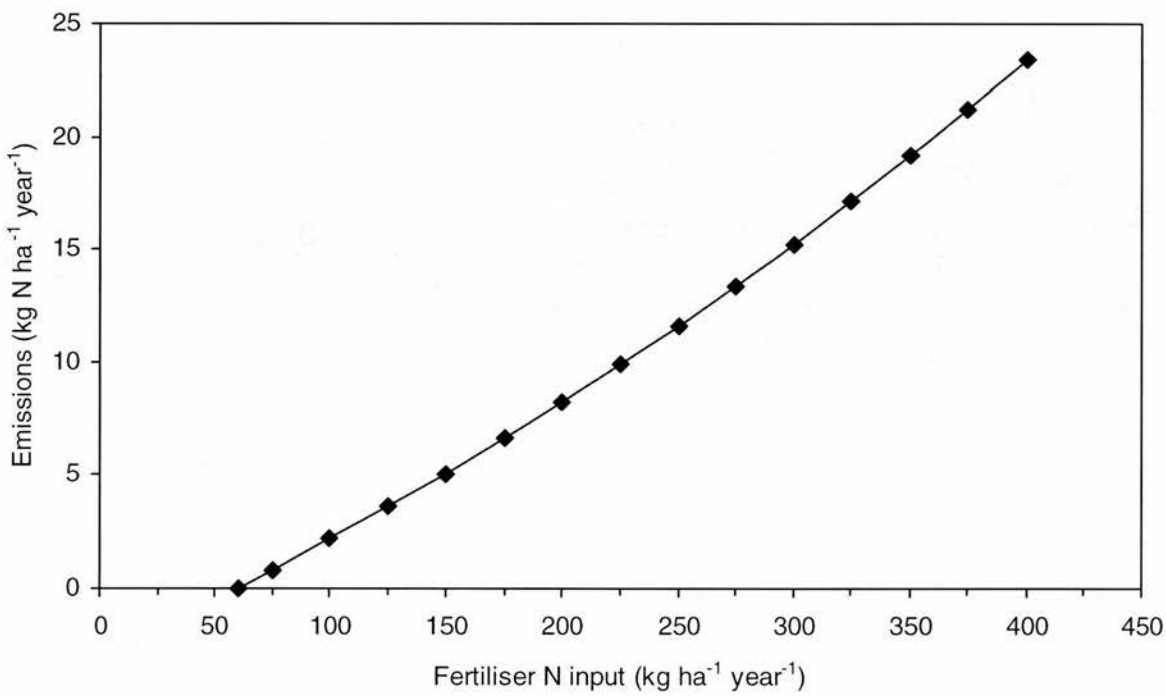


Figure 3.6. Ammonia emissions from grazed grassland for cattle and sheep (after Pain *et al.*, 1997)

However, the BBSRC inventories (1997a, 1997b) use only one point on the curve shown in Figure 3.6. to derive their average UK grazing emissions from dairy cows, rather than applying the whole curve to the range of fertiliser N rates applied by UK farmers. Because of the non-linearity of the curve, this leads to an underestimate of NH_3 emissions from grazing livestock.

Regarding absolute emission estimates from grazing dairy cows, the latest BBSRC (1997b) loss rate results in emissions of 2.4 kg N per cow, which is significantly lower than the 4.7, 4.3 and 3.2 kg N estimated by Asman (1992b), ECETOC (1994) and TFEI (1996). This is not only a function of the smaller loss rate, but also related to the larger amount of N excretion assumed by the latter 3 studies, despite the shorter grazing period. If the loss rate of 5.6% estimated by BBSRC (1997b) was applied to the available N from a dairy cow as estimated by Asman, the loss would be 3.3 kg N. Alternatively, an 8% loss rate applied to a BBSRC (1997b) type dairy cow would result in a loss of 3.4 kg N. However, considering that grazing emissions contribute only approximately 10-14% of the total emissions from dairy cows, it is clear that this is not the main source of uncertainty.

For emissions from housing and manure storage, the studies quoted in Table 3.3. show a range of loss rates between 13.2% (Asman, 1992b) and 23.3% (BBSRC, 1997b) of the N excreted indoors. In absolute terms, emissions from housing and storage range between 8.3 (BBSRC, 1997a) and 12.7 kg N (BBSRC, 1997b) per dairy cow. Discounting the earlier BBSRC estimate of 8.3 kg N as outdated, the absolute emission estimates are from housing and storage for the studies are surprisingly similar, considering the much larger range of the relative loss rates. This is partially due to the fact that Asman (1992b), whose dairy cows have the highest N pool available for emission, estimates the smallest loss rates, whereas the BBSRC (1997b) study with the smallest N pool available for NH_3 volatilisation estimates the highest loss rates during housing and storage. In total, housing and storage emissions contribute about 32% (Asman, 1992b) to 53% (BBSRC, 1997b) of the total emissions, with the other studies in the middle of this range at 38% (ECETOC, 1994) to 44% (TFEI, 1996; BBSRC, 1997a).

Regarding emissions from the landspreading of manures from dairy cows, the range of loss rates is between 16.5% (BBSRC, 1997a) and 25% (Asman, 1992b) of the available N, with absolute losses of between 8.8 kg N (BBSRC, 1997b) and 17.4 kg

N (Asman, 1992b). Again, the much larger differences in the absolute emissions from landspreading are due to the much larger amount of N available for volatilisation (see row "N available before spreading" in Table 3.3.). For instance, Asman's (1992b) estimate of N available before landspreading is nearly double that of the BBSRC (1997b) study. In total, landspreading emissions are considered the largest component emission source for dairy cattle by most studies at 37% (BBSRC, 1997b) to 53% (Asman, 1992b) of the total annual NH_3 emission per animal.

b) Other cattle/beef cattle

The category 'other cattle' summarises all cattle which are not dairy cows, i.e. beef cows, bulls and young cattle (dairy replacements and beef cattle), including calves. There is a large variation in age, liveweight, N input to feed, housing & storage conditions, and duration of the grazing season, all accumulated into one group. (For instance, some beef cattle are reared entirely indoors ("barley beef"), others spend a large proportion of their lives outdoors.) The BBSRC (1997a and 1997b) inventories provide 8 subclasses to take account of the different parameters influencing NH_3 emissions from this diverse group. The other study included in Table 3.3. and Figures 3.4. and 3.5. above for the category "other cattle" is the TFEI (1996) inventory.

The amount of N available for volatilisation of NH_3 from cattle excreta appears to be rather uncertain if one compares the estimates by the BBSRC group (1997a and b), 25.9 kg and 26.8 kg per animal respectively, with the estimate of 50 kg per animal by the TFEI study (1996). A main reason for this is again likely to be found in the N content of the animal feed: the BBSRC studies (1997a, 1997b) base their estimates at a very low level of approx. 75 kg N ha^{-1} fertiliser input, whereas TFEI (1996) appears to assume the same N input level for all cattle. Regarding relative losses from the initial N pool, TFEI (1996) takes the same values as for dairy cows, stating that losses from straw-based FYM and slurry were assumed the same in their simple methodology. They acknowledge, however, that in reality FYM emissions are lower than slurry emissions during housing, but higher during storage. Regarding overall losses from the N available from excretion, the studies do not appear very different at a first glance: The rather low value of 13.9% in the earlier BBSRC study (1997a) was updated to 19.4% in the newer version (1997b), which is not too different from the 23.5% assumed in the TFEI (1996) study. In absolute terms (5.2 kg N and 11.74 kg N per animal respectively), the main difference can be attributed to the figures assumed for total N excreted.

More differences become obvious when the component emission estimates are compared: due to the low N input assumed, the loss rate for cattle grazing is very low in the BBSRC studies (see Table 3.3.; Figures 3.4. and 3.5.), although the estimate of 0.7% in the earlier BBSRC spreadsheet (1997a) was revised upwards to 2.8% (BBSRC, 1997b). This still only accounts for only 7.6% of the total emissions (BBSRC, 1997b). In contrast, the corresponding rate in the TFEI (1996) inventory is nearly twice as high, i.e. grazing accounts for 13.6% of the total emissions).

Housing and storage losses are rather higher in the more recent BBSRC inventory (1997b), at 30.8% of the N excreted during housing, compared with 19.1% in the earlier version and 18% in the TFEI (1996) study. Spreading losses between the two main groups of studies vary again quite considerably, between 13.8% (BBSRC 1997a and 1997b) and 20% (TFEI, 1996) of the N left in the manure before spreading. The lower estimate for landspreading emissions from beef cattle (17%) in the BBSRC (1997b) study, compared with dairy cows (21%) is caused by the different housing systems assumed for the 2 categories. Beef cattle in the UK are mainly housed on straw (60%), whereas the majority of dairy cows (85%) produce slurry during the housing period. This leads to higher housing and lower spreading emissions for beef cattle in the BBSRC estimates. Compared with dairy cows, the uncertainties encountered for "other cattle" are rather larger, especially when the component emission estimates are considered more closely.

c) Average cattle

In this category, 7 studies are compared in Table 3.3. and Figures 3.4. and 3.5. above: Buijsman *et al.* (1987), Jarvis and Pain (1990) and the earlier BBSRC study (1997a) are now considered outdated or were revised recently. Therefore the focus is on the latest BBSRC study (1997b), Sutton *et al.* (1995), TFEI (1996) and ECETOC (1994) in the following paragraphs.

Sutton *et al.* (1995), TFEI (1996) and BBSRC (1997b) agree more or less on the average N excretion rate per animal, with estimates of 60-65 kg N animal^{-1} year^{-1}. It should be noted that the BBSRC estimate of N excretion for average cattle has increased considerably from 43.8 kg N animal^{-1} year^{-1} since Jarvis and Pain (1990).

The figure provided by the ECETOC report (1994) is higher at 80.7 kg N per animal, which is likely to be related to the assumed higher N input to feed in this study. Another explanation may be found in the cattle categories used in the ECETOC study: The three classes are cattle over 2 years, cattle between 1-2 years and cattle under 1 year old. Average emissions for the 3 categories are 32.5 kg, 19.7 kg and 11.2 kg N per animal respectively. The estimates for dairy cattle discussed in Section a) above are taken to be valid for all cattle over 2 years old. In reality, nearly half of the cattle counted in this category are suckler cows, heifers, replacement dairy cattle and beef cattle, which emit less NH_3 than dairy cows. Furthermore, the estimates for 1-2 year old cattle are based on a N fertiliser application rate of 300 kg N ha^{-1}, which is an overestimate compared with average UK conditions (Burnhill *et al.*, 1997). By re-adjusting the animal numbers for the cattle subclasses and using lower N input estimates, the total cattle emissions in the ECETOC report are likely to decrease by a substantial amount and be more in line with the other estimates in Tables 3.1. and 3.3. above.

The relative amount volatilised from the initially available N ranges between 15.9% (BBSRC, 1997b) and 28.4% (ECETOC, 1994), with Sutton *et al.* (1995) and TFEI (1996) estimating rates of 25.8% and 23.5%, respectively. The emission estimates vary between 9.9 kg and 22.9 kg N per animal, again with the BBSRC (1997b) estimate at the lower and the ECETOC (1994) estimate at the higher end, and TFEI (1996) and Sutton *et al.* (1995) in the middle (see Table 3.3.).

Comparing the component emission factors in Table 3.3., the BBSRC study (1997b), TFEI (1996) and Sutton *et al.* (1995) agree roughly on the magnitude of the three main components to the total emissions, with grazing emissions as the smallest component (12%, 13% and 17% of the total emissions per animal), followed by landspreading emissions (39%, 43% and 37%), and with housing and storage emissions providing the largest component (48%, 44% and 46%). The ECETOC (1994) estimates are similar for grazing emissions (14% contribution), but landspreading emissions are the largest component emission source at 49%, followed by housing and storage emissions at 37%.

It appears that an agreement has been reached between the most recent studies regarding the annual excretion rate of 60-65 kg for average cattle. There is less consensus regarding the component emission estimates. On an overall basis, however, the TFEI (1996) estimates are best supported by the literature and have been chosen here as the best available values according to present knowledge.

3.2.2. Ammonia emission estimates for sheep

According to the latest emission inventories (see Table 3.4., Figures 3.7. and 3.8.), NH_3 emissions from sheep contribute between 12.7 and 34.3 kt N, i.e. between 6.2 and 11.2 % of the UK NH_3 emissions from the main livestock sources. This is a difference of 21.6 kt N between the highest and lowest estimates. As is the case for the contribution from cattle, the BBSRC estimates (1997a, 1997b) provide the lowest estimates at 12.7 and 12.9 kt N, respectively. The DoE (1995) inventory figures are of the same order at 15.8 kt N, followed by TFEI (1996) at 22.9 kt and ECETOC (1994) at 32.9 kt N. (NB: The original ECETOC total estimate for sheep for 1990 was much larger (46.5 kt), due to an error in the number of sheep for the UK.) Sutton *et al.*'s (1995) estimate is nearly 3 times as high as the BBSRC estimates at 34.3 kt N. Although the absolute uncertainty range is much smaller for sheep (21.6 kt N) than for cattle (168.8 kt N), the relative range of uncertainty is similar, with the highest estimate about 2.5-3 times higher than the lowest. It is therefore imperative to examine the individual emission estimates per animal more closely in order to narrow this uncertainty range.

In contrast to other livestock types, the largest contribution to sheep emissions is from grazed pastures (Table 3.4.; Figures 3.7. and 3.8.). Sheep spend most of the year outside, with only about one month's housing for lowland sheep in late winter under average UK conditions (BBSRC, 1997b). Upland sheep are estimated to be outdoors all year on average in the UK (BBSRC, 1997b). Therefore, housing, storage and landspreading emissions are much smaller than for cattle.

The studies in Table 3.4. agree remarkably well on the relative amount of NH_3 lost from the total available N excreted per sheep - between 5.5% (ECETOC, 1994) and 8.3% (BBSRC, 1997a). The absolute amounts lost per sheep, however, show large differences: For ewes without lambs, the range of annual NH_3 emission estimates is 0.5 - 1.2 kg N per animal. For combined emissions from ewes and their lambs, the range is 0.6 - 1.54 kg N per ewe. The method of including the emissions from lambs with their mothers has several flaws. The main difficulty with these aggregated emissions arises from the fact that the average lambing rate per ewe is different for different regions and countries, mainly depending on the amount and quality of food available to the ewes at critical times of the reproduction cycle. Sheep kept on fertile pastures in lower lying areas generally have a higher lambing rate than upland sheep, which have to subsist on a much poorer diet.

The ECETOC study assumes an average of 2 lambs per ewe, which is higher than the mean UK lambing rate of 1.1-1.2 (Webb, 1995). Thus, applying the ECETOC (1994) estimate to the UK ewe population would result in an overestimate of total sheep emissions. The TFEI study (1996) assumes a lambing rate of 1-1.5, which is closer to the UK average conditions. To facilitate comparisons, estimates for ewes without lambs were derived from combined data for ewes and lambs (ECETOC, 1994), and emissions for ewes and lambs were aggregated (BBSRC 1997a, 1997b) and included in Table 3.4. In general, it makes more sense to keep ewes and lambs separate, if suitable data are available for both categories in the agricultural census. This is especially true for a spatially distributed inventory, where regional differences in lambing rates may be significant.

The largest difference between the inventories discussed here is the total N excretion rate per sheep, with the lowest estimates at ~6kg N per ewe (BBSRC, 1997a and 1997b) and the highest estimate at 17.7 kg N per ewe (derived from ECETOC, 1994; 23 kg N per ewe including twin lambs). The low BBSRC (1997a, 1997b) estimates contrast sharply with their predecessor study by Jarvis and Pain (1990), who quote N excretion rates of 23.7 kg N in their Table 3 (see also Sutton *et al.*, 1995). The TFEI

(1996) estimate is also at the higher end with 20 kg for a ewe with 1-1.5 lambs. Neither of the 2 extremes agrees with the ADAS estimate of 8-10 kg per ewe for UK conditions (Webb, 1995). Another recent UK study by Orr *et al.* (1995) shows the dependence of sheep N excretion rates on N input to pastures: non-lactating ewes excreted about 14.5 kg N annually on unfertilised grass, 16.1 kg N on an unfertilised grass-clover pasture, and 23.2 and 23.5 kg N respectively on grass with 420 kg N fertiliser input per hectare and unfertilised clover. The sheep under investigation in this study were Finn Dorset ewes with an average liveweight of 62 kg, which is about the average for a medium weight breed (see Chapter 2). All of these values are well above the estimates suggested by the BBSRC (1997a, 1997b) inventories. The highest values shown in Table 3.4. appear to be more suitable for sheep on highly fertilised pastures, and their use for emission inventory purposes would lead to an overestimate for most UK flocks.

Summarising, it can be stated that NH_3 emission estimates for sheep vary widely between the recent studies described and analysed in this section, regarding the amount of N excreted and thus available for volatilisation, as well as the component emission estimates. The following is suggested as a best estimate: A nitrogen excretion rate of 15-17 kg per ewe appears well supported by the literature (Orr *et al.*, 1995; TFEI, 1996; Sutton *et al.*, 1995). Grazing losses may be estimated within a range of 4-8%, with a conservative best estimate of 5%. Housing losses for a period of 30 days for lowland ewes (51.5% of all ewes in the UK; BBSRC, 1997b) are estimated at 10 %of the available N following TFEI (1996), with an uncertainty range of 5-15%. For storage emissions, the loss rate from cattle in TFEI (1996) was scaled for sheep, resulting in a source strength estimate of 2%. Best estimates for landspreading emissions from sheep were set at 10%, again following TFEI (1996). This results in an average emission of 0.92 kg N per ewe, which is close to the TFEI estimate of 1.1 kg N for one sheep and 1-1.5 lambs.

Table 3.4. Ammonia emissions from sheep according to different studies showing the N flow: N excretion rates, absolute and relative component losses (units: *italic*: % volatilisation rate; **bold**: emissions in kg NH_3-N animal^{-1} year^{-1}; normal: available N in kg NH_3-N animal^{-1} year^{-1}).

	ECETOC	BBSRC	BBSRC	BBSRC	BBSRC	TFEI	ECETOC	best estimate
	1994	1997a	1997b	1997a	1997b	1996	1994	1998
	ewe	ewe	ewe	ewe+lamb	ewe+lamb	ewe+1-1.5lambs	ewe+2lambs	ewe
total N excreted	17.7	5.86	6.03	7.42	7.77	20	23	16
field - % avail. N	93%	92%	92%	94%	94%	90%	93%	95%
field - available N	16.54	5.38	5.53	6.94	7.27	18	21.5	15.2
field loss rate	*4.6%*	*6.9%*	*8.0%*	*6.8%*	*7.7%*	*4.0%*	*4.6%*	*5%*
grazing emission	**0.76**	**0.37**	**0.44**	**0.47**	**0.56**	**0.72**	**0.99**	**0.76**
housing - avail. N	1.16	0.48	0.5	0.48	0.5	2	1.5	0.8
housing loss rate	*12.1%*	*16.7%*	*2.0%*	*16.7%*	*2.0%*	*10.0%*	*12.0%*	*10%*
housing emission	**0.14**	**0.08**	**0.01**	**0.08**	**0.01**	**0.2**	**0.18**	**0.08**
N avail. before storage	1.02	0.4	0.49	0.4	0.49	1.8	1.32	0.72
storage loss rate	*0.0%*	*1.0%*	*1.0%*	*1.0%*	*1.0%*	*0.0%*	*0.0%*	*2%*
storage emission	**0**	**0.004**	**0.005**	**0.004**	**0.005**	**0**	**0**	**0.01**
N avail. before spreading	1.02	0.396	0.485	0.396	0.485	1.8	1.32	0.71
spreading loss rate	*27.9%*	*7.6%*	*6.2%*	*7.6%*	*6.2%*	*10.0%*	*28.0%*	*10%*
spreading emission	**0.285**	**0.03**	**0.03**	**0.03**	**0.03**	**0.18**	**0.37**	**0.07**
N avail. to plants	0.735	0.366	0.455	0.366	0.455	1.62	0.95	0.64
total emission	**1.185**	**0.484**	**0.485**	**0.584**	**0.605**	**1.1**	**1.54**	**0.92**
% of total N excreted volatilised	*6.7%*	*8.3%*	*8.0%*	*7.9%*	*7.8%*	*5.5%*	*6.7%*	*5.8%*

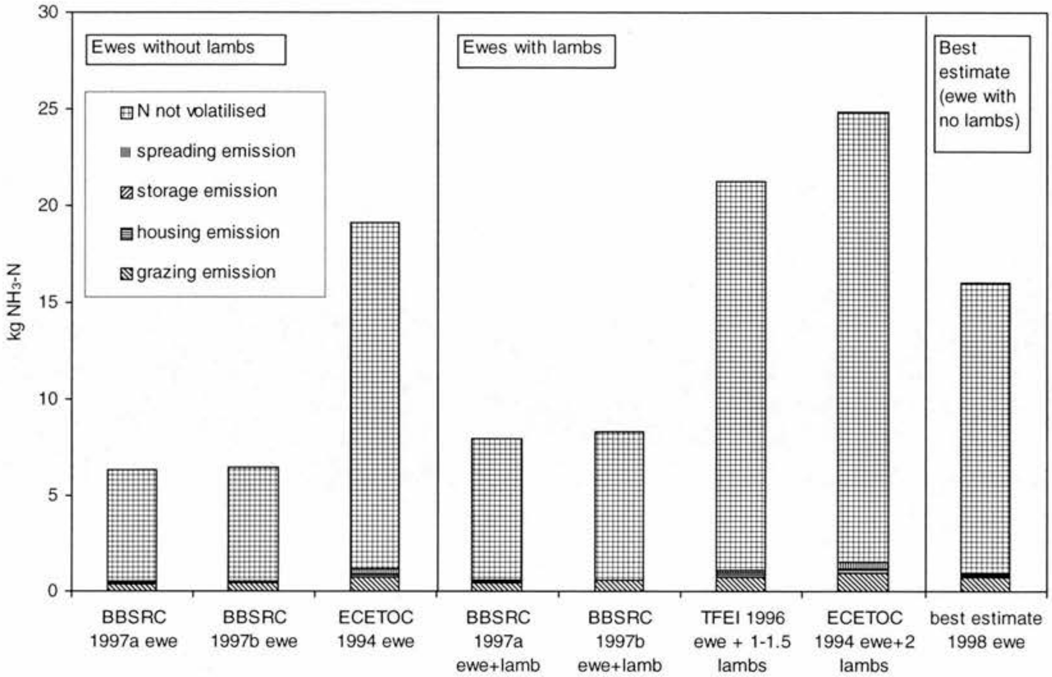


Figure 3.7. Component NH_3 losses from sheep (grazing, housing, manure storage and landspreading of manures) in kg NH_3-N animal^{-1} year^{-1} (derived from Table 3.4. above).

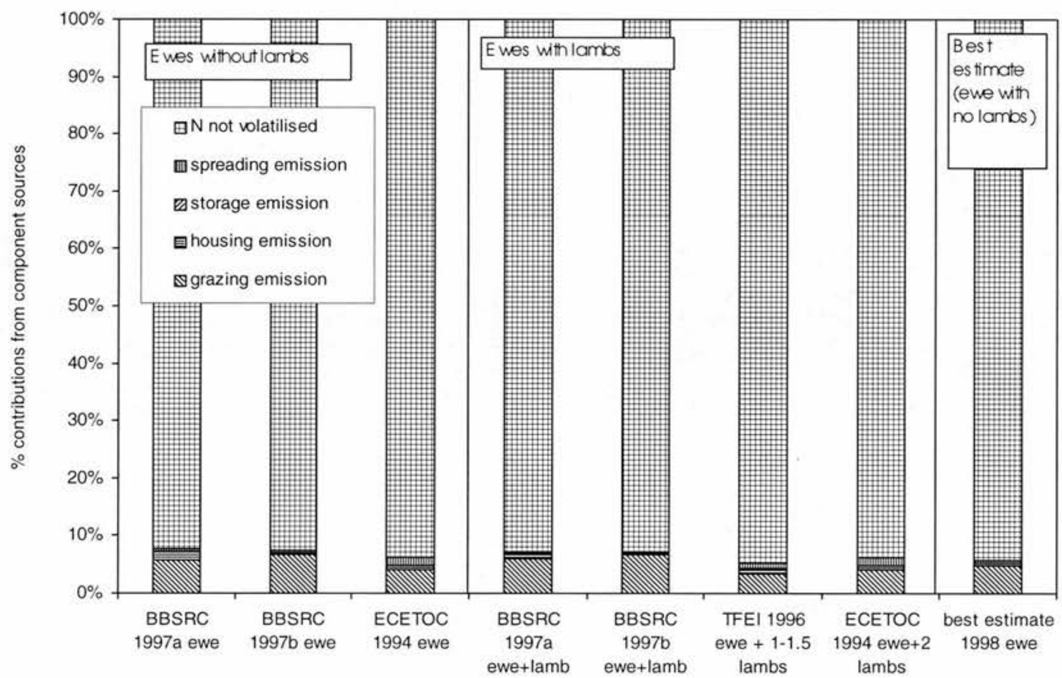


Figure 3.8. Component NH_3 losses from sheep (grazing, housing, manure storage and landspreading of manures) as % of N excreted (derived from Table 3.4. above).

3.2.3. Ammonia emission estimates for pigs

According to the inventories discussed here, NH_3 emissions from pigs amount to 22.2 - 32.2 kt N in the UK for 1996 (Table 3.1.). The BBSRC inventories provide the lowest estimates again, with 22.2 kt N in their earlier version (BBSRC, 1997a) and 25.7 kt N in the latest estimate (BBSRC, 1997b). The low estimate by DoE (1995) is heavily influenced by the earlier BBSRC study (1997a). The latest BBSRC estimate (1997b) is very similar to the ECETOC estimate (1994; see Table 3.5.), while the TFEI (1996) inventory and Sutton *et al.* (1995) provide the highest numbers. Overall pig emission estimates are equivalent to a contribution of 7.2-13.2% to the main livestock emissions described in Table 3.1. and Figures 3.1 and 3.2. above.

The uncertainty range regarding pig emissions appears to be very small compared with cattle and sheep (see Figures 3.1. and 3.2.), if just the summarised total emissions are considered. However, a closer look at the calculations and their underlying assumptions (see Table 3.5.; Figures 3.9. and 3.10.) show large differences in the estimated component emissions per pig.

For (indoor) sows, the estimated annual N excretion rates in the latest inventories are very similar, varying between 31.2 kg (BBSRC, 1997b) and 36.0 kg (TFEI, 1996). The ECETOC (1994) and TFEI (1996) estimates include emissions from piglets under 20 kg with the sows' emissions, while the BBSRC studies (1997a, 1997b) appear to account for piglets separately. The TFEI (1996) study also adds 4 kg N for young sows (equivalent to 0.3 young sows per breeding sow) to the original 32 kg N per sow. The estimate of 15 kg N excreted per sow in the earlier BBSRC inventory (BBSRC, 1997a) is now assumed to be outdated and has been doubled in the most recent version (BBSRC, 1997b).

Despite this agreement on the average N excretion rates per sow, the NH_3 emission estimates vary considerably, between 3.8 kg (12% of excreted N; BBSRC, 1997b) and 13.5 kg N per sow (38% of excreted N; TFEI, 1996), with the ECETOC (1994) estimate also towards the higher end of the range at 9.8 kg (30% of the excreted N). While the 2 latter studies agree more or less on emissions from housing and storage (7.9 and 8.6 kg respectively), the BBSRC study (1997b) provides the lowest estimate at 2.8 kg. For manure spreading, the emission estimates show again a wide range: between 3.5% and 20% of the N available after housing and storage is estimated to volatilise, with both the ECETOC (1994) and BBSRC (1997 a and 1997b) studies at the lower end of the range. The low landspreading emissions per sow in the BBSRC studies (1997a and 1997b) are partly due to the authors not weighting landspreading losses according to animal weight or excretal output. This results in increasing loss rates with decreasing animal weight. Thus, manure spreading emissions from sows are underestimated, while those from young pigs are overestimated, compared with average pigs in the BBSRC studies. The average emissions from landspreading of pig manures (13%) estimated by the BBSRC studies (1997a and 1997b) is, however, still on the low side, compared with now generally accepted loss rates of about 20% (Sutton *et al.*, 1995). The large overall difference in estimated NH_3 emissions per sow by the different studies does not have a large effect on the summarised emission totals per pig. This is because sows contribute only a small fraction (11% in 1996) of the total UK pig population.

Fattening pig emission estimates, on the other hand, agree very closely between the different studies, regarding average N excretion rates as well as total NH_3 losses (see Table 3.5.). BBSRC (1997b) divide their fattening pigs into 3 main classes according to weight, with separate estimates for each class (< 20 kg, 20-110 kg, > 110 kg). In the following, their two heavier categories are compared with the TFEI (1996) and ECETOC (1994) average fattening pigs.

The N excretion rates per fattening pig agree very well between the studies, with the BBSRC (1997b) inventory's lighter and heavier fattener classes providing the lowest and highest estimates, and TFEI (1996) and ECETOC (1994) within this range of 10.9-14.9 kg N pig^{-1} $year^{-1}$. As regards total losses of N as NH_3 per pig, all studies agree more or less with loss rates of 30-42% of the total available N (3.8-6.3 kg NH_3-N pig^{-1} $year^{-1}$). Emissions from housing and storage range from 22-36% of the total excreted N, and spreading emissions from 7-16%. If spreading emissions are calculated as % rate volatilisation of the N available in pig manure after housing and storage losses are accounted for, the loss rates are 5-20% (see Table 3.5.). Sutton *et al.* (1995) reject the lower limit of this range (ECETOC, 1994) as too conservative compared with general experimental evidence (e.g. Jarvis and Pain 1990; Asman, 1992b), which points to emission rates of 18-25%, respectively. If this was corrected, the ECETOC (1994) emission estimate per fattener would agree even more closely with the other three estimates.

As a best estimate according to the information available at present the TFEI (1996) values were chosen for average pigs. Their N excretion rates as well as their component emission source strength estimates agree favourably with most other estimates in Table 3.5. Compared with the latest BBSRC study (1997b), the housing losses estimated by TFEI (1996) are slightly more conservative. Regarding landspreading emissions, the TFEI estimates are more reliable in terms of the basic literature (see e.g. Sutton *et al.*, 1995) than the BBSRC estimates.

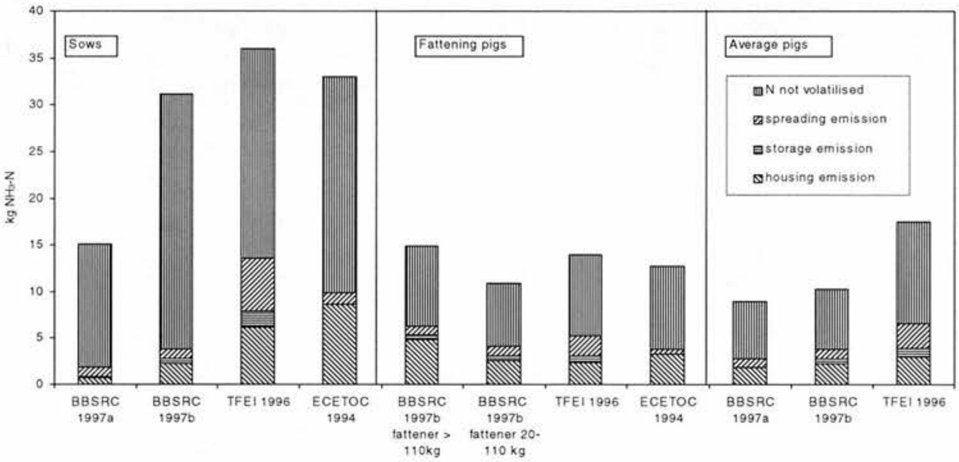


Figure 3.9. Component NH_3 losses from pigs (housing, manure storage and landspreading of manures) in kg NH_3-N animal-1 year-1 (derived from Table 3.5. above).

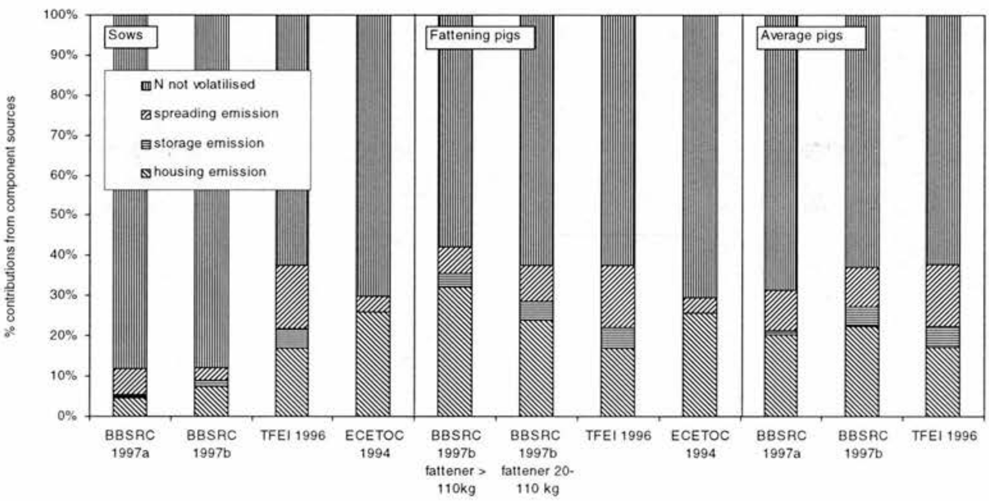


Figure 3.10. Component NH_3 losses from pigs (housing, manure storage and landspreading of manures) as % of N excreted (derived from Table 3.5. above).

Table 3.5. Ammonia emissions from pigs according to different studies showing the N flow: N excretion rates, absolute and relative component losses (units: *italic*: % volatilisation rate; **bold**: emissions in kg NH_3-N animal^{-1} year^{-1}; normal: available N in kg NH_3-N animal^{-1} year^{-1}).

	BBSRC 1997a Indoor sow	BBSRC 1997b Indoor sow	TFEI 1996 sow	ECETOC 1994 sow/boar	BBSRC 1997b fattener > 110 kg	BBSRC 1997b fattener 20-110 kg	TFEI 1996 fattener	ECETOC 1994 fattener	BBSRC 1997a avg. pig	BBSRC 1997b avg. pig	TFEI 1996 avg. pig
total N excreted	15.1	31.2	36	33	14.9	10.9	14	12.8	8.9	10.3	17.4
field - % avail. N	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
field - available N	0	0	0	0	0	0	0	0	0	0	0
field loss rate	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.00%*
grazing emission	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.02**	**0.1**	**0**
housing - avail. N	15.1	31.2	36	33	14.9	10.9	14	12.8	8.9	10.3	17.4
housing loss rate	*4.6%*	*7.4%*	*17.0%*	*26.1%*	*32.2%*	*23.9%*	*17.0%*	*25.8%*	*20.2%*	*22.3%*	*17%*
housing emission	**0.7**	**2.3**	**6.1**	**8.6**	**4.8**	**2.6**	**2.4**	**3.3**	**1.8**	**2.3**	**3.0**
N avail. before storage	14.4	28.9	29.9	24.4	10.1	8.3	11.6	9.5	7.1	8	14.4
storage loss rate	*0.7%*	*1.7%*	*6.0%*	*0.0%*	*5.0%*	*6.0%*	*6.0%*	*0.0%*	*1.4%*	*6.3%*	*6%*
storage emission	**0.1**	**0.5**	**1.79**	**0**	**0.5**	**0.5**	**0.7**	**0**	**0.1**	**0.5**	**0.9**
N avail. before spreading	14.3	28.4	28.09	24.4	9.6	7.8	10.9	9.5	7	7.5	13.6
spreading loss rate	*7.0%*	*3.5%*	*20.0%*	*5.0%*	*10.4%*	*12.8%*	*20.0%*	*5.3%*	*12.9%*	*13.3%*	*20%*
spreading emission	**1**	**1**	**5.6**	**1.2**	**1**	**1**	**2.2**	**0.5**	**0.9**	**1**	**2.7**
N excreted during housing & not volatilised	13.3	27.4	22.5	23.2	8.6	6.8	8.7	9	6.1	6.5	10.9
total emission	**1.8**	**3.8**	**13.5**	**9.8**	**6.3**	**4.1**	**5.3**	**3.8**	**2.8**	**3.9**	**6.5**
% of total N excreted	*11.9%*	*12.2%*	*37.6%*	*29.8%*	*42.3%*	*37.6%*	*37.6%*	*29.7%*	*31.7%*	*37.9%*	*37.6%*

3.2.4. Ammonia emission estimates for poultry

The term 'poultry' is applied here to a range of domestic birds, which are kept in the UK under a variety of different management practices. This includes chickens (treated separately for egg and meat production), ducks, geese and turkeys. In the following paragraphs, NH_3 emission estimates from different inventories are critically reviewed and compared (see Table 3.6.), mainly for laying hens, broilers and turkeys, which are the most frequent species in the UK. Poultry emissions for the UK have been estimated at 21.9-44.2 kt by recent studies (see Table 3.1., Figures 3.1. and 3.2. above), contributing 6-21% of the total NH_3 emissions from the main livestock categories.

Considering laying hens first, the BBSRC (1997b), TFEI (1996) and ECETOC (1994) inventories' estimates of total excreted N and total NH_3 emissions are all within very narrow ranges, at 0.68-0.8 and 0.31-0.33 kg N bird^{-1} year^{-1}, respectively. This constitutes relative loss rates of 39-46% of the total excreted N. The main differences between the studies emerge only when the component emission rates are considered. Whereas the BBSRC (1997b) and TFEI (1996) loss rates for housing & storage (27% and 24% of the total excreted N) and manure spreading (17.5 and 15% of the total excreted N) are very similar, ECETOC (1994) disagree. They estimate that only 9% of the total excreted N volatilise during housing and storage, and 34% (of the total excreted N) during landspreading.

For broilers, there is again reasonable agreement between the BBSRC (1997b) and TFEI (1996) inventories (see Table 3.6.), whereas the ECETOC (1994) estimate is differing to a larger degree: its total N excretion rates are only about half of the other 2 inventories, and in terms of absolute total emission estimates even less at about one third of the other 2 inventories.

There are only two inventories with detailed emission estimates for turkeys available, BBSRC (1997b) and TFEI (1996). The 'other poultry' estimate of the latter is based on turkeys. Despite very similar total emission estimates at 0.81 kg (BBSRC, 1997b) and 0.76 kg (TFEI, 1996), respectively, the studies differ greatly in the derivation of these figures. The much larger total N excretion rates of TFEI (1996) are

Table 3.6. Ammonia emissions from poultry according to different studies showing the N flow: N excretion rates, absolute and relative component losses (units: *italic*: % volatilisation rate; **bold**: emissions in kg NH_3-N animal^{-1} year^{-1}; normal: available N in kg NH_3-N animal^{-1} year^{-1}).

	BBSRC 1997a layer	BBSRC 1997b layer	TFEI 1996 layer	ECETOC 1994 layer	BBSRC 1997a broiler	BBSRC 1997b broiler	TFEI 1996 broiler	ECETOC 1994 broiler	BBSRC 1997b turkey	TFEI 1996 turkey
total N excreted	0.63	0.68	0.8	0.766	0.609	0.837	0.6	0.348	1.212	2.0
field - % avail. N	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
field - available N	0	0	0	0	0	0	0	0	0	0
field loss rate	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*
grazing emission	**0**	**0.013**	**0**	**0**	**0**	**0**	**0**	**0**	**0.011**	**0**
housing - avail. N	0.63	0.68	0.8	0.766	0.609	0.837	0.6	0.348	1.212	2.0
housing loss rate	*31.1%*	*26.0%*	*20.0%*	*9.1%*	*9.0%*	*15.5%*	*20.0%*	*15.8%*	*53.8%*	*20.0%*
housing emission	**0.196**	**0.177**	**0.16**	**0.07**	**0.055**	**0.13**	**0.12**	**0.055**	**0.652**	**0.4**
N avail. before storage	0.434	0.503	0.64	0.696	0.554	0.707	0.48	0.293	0.56	1.6
storage loss rate	*1.2%*	*0.9%*	*4.7%*	*0.0%*	*0.9%*	*0.6%*	*2.1%*	*0.0%*	*0.0%*	*3.1%*
storage emission	**0.0054**	**0.0043**	**0.03**	**0**	**0.0051**	**0.0041**	**0.01**	**0**	**0**	**0.05**
N avail. before spreading	0.4286	0.4987	0.61	0.696	0.5489	0.7029	0.47	0.293	0.56	1.55
spreading loss rate	*25.7%*	*23.9%*	*19.7%*	*37.6%*	*15.3%*	*14.7%*	*19.1%*	*6.8%*	*25.5%*	*20.0%*
spreading emission	**0.11**	**0.119**	**0.12**	**0.262**	**0.084**	**0.103**	**0.09**	**0.02**	**0.143**	**0.31**
N excreted during housing & not volatilised	0.3186	0.3797	0.49	0.434	0.4649	0.5999	0.38	0.273	0.417	1.24
total emission	**0.3114**	**0.3133**	**0.31**	**0.332**	**0.1441**	**0.2371**	**0.22**	**0.075**	**0.806**	**0.76**
% of total N excreted volatilised	*49.4%*	*46.1%*	*38.8%*	*43.3%*	*23.7%*	*28.3%*	*36.7%*	*21.6%*	*66.5%*	*38.0%*

compensated for in their much smaller relative loss rate from housing and storage (see Table 3.6., Figures 3.11. and 3.12.).

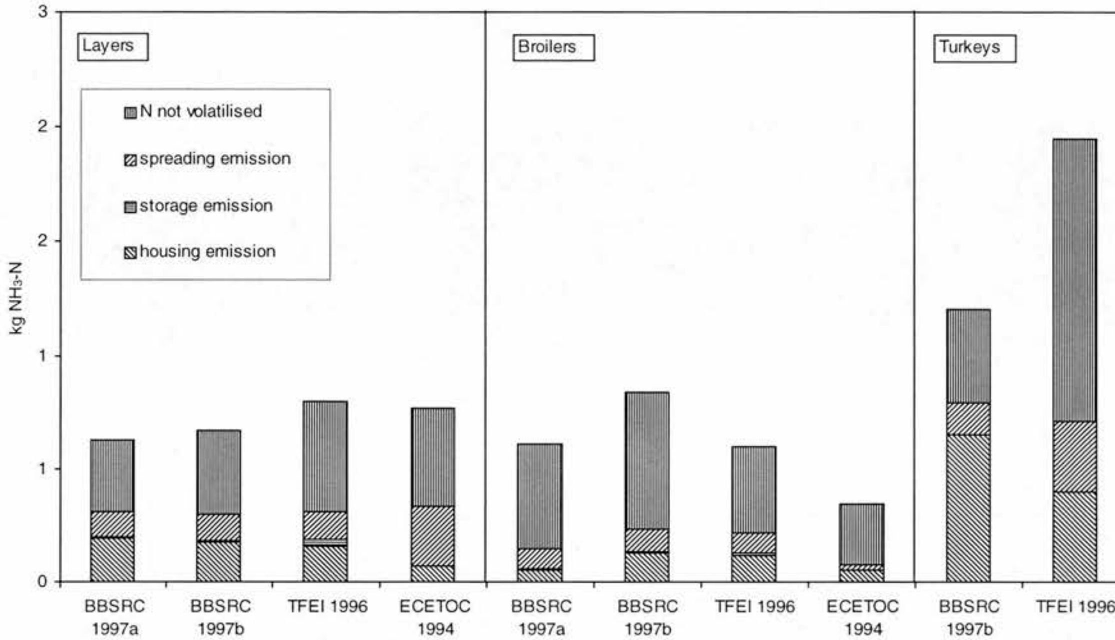


Figure 3.11. Component NH_3 losses from poultry (housing, manure storage and landspreading of manures) in kg NH_3-N animal-1 year-1 (derived from Table 3.6. above).

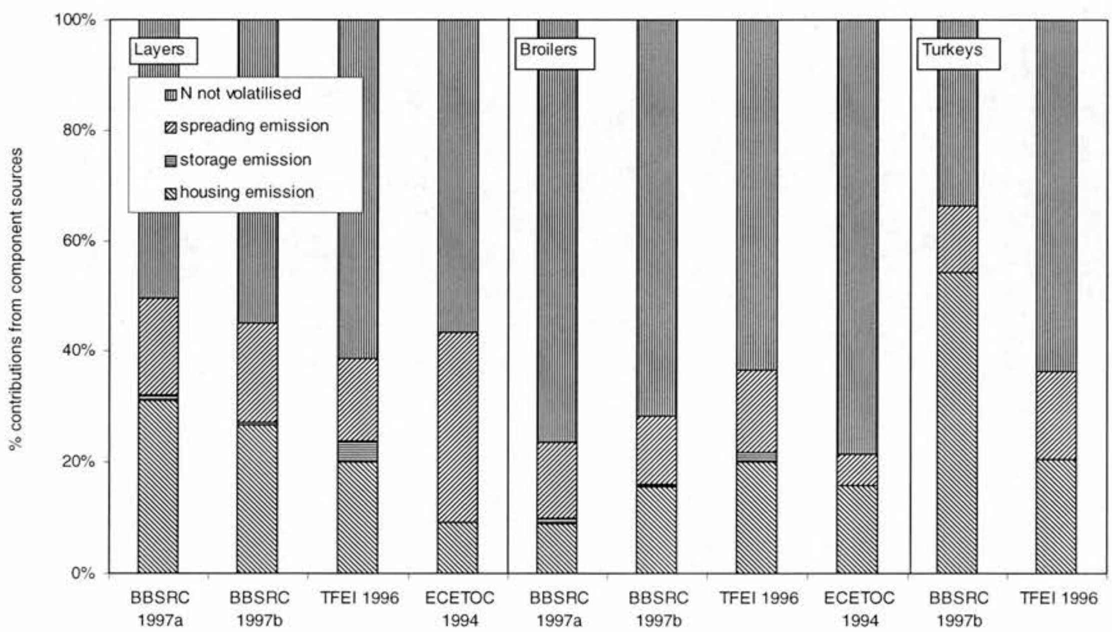


Figure 3.12. Component NH_3 losses from poultry (housing, manure storage and landspreading of manures) as % of N excreted (derived from Table 3.6. above).

Considering the comparisons above, it appears that the original large range of UK poultry emissions of 21.9-44.2 kt N can be reduced to a more realistic one of 37.2-44.2 kt N, with the ECETOC (1994) estimate being discounted for its rather too low broiler emissions. The older BBSRC (1997a) estimate (29.1 kt) and the DoE (1995) estimate (26.9 kt N), which is closely linked to it, can now be considered as outdated by the latest estimates (BBSRC, 1997b).

3.2.5. Ammonia emission estimates for other livestock

Cattle, sheep, pigs and poultry together contribute nearly all the NH_3 emissions from agricultural livestock in the UK. There are, however, minor contributions from other livestock, such as horses, goats and farmed deer.

Emissions from horses are dealt with in Section 3.4., as a substantial part of the UK horse population are not counted as agricultural livestock. Emissions from goats can be treated as equivalent to emissions from sheep, due to the relatively small number of goats in the UK (81,000 animals in 1996; BBSRC, 1997b), the similarity in husbandry practice, and a lack of specialised studies.

There are no dedicated studies available either for NH_3 emissions from farmed deer. Estimates were made for wild deer by Sutton *et al.* (1995), who adjusted the sheep emission estimate by Jarvis and Pain (1990) for the higher metabolic rate of red deer (equivalent to 2.5 sheep). This resulted in an emission estimate of 0.9 kg N animal^{-1} year^{-1}. Updating the original sheep estimate to the latest BBSRC (1997b) figures, a new estimate for deer can be derived at 1.23 kg N animal^{-1} year^{-1}. The estimates for farmed deer in the BBSRC study (1997b) were also derived from sheep emission estimates (Pain *et al.*, 1998), resulting in 0.85 kg animal^{-1} year^{-1}. These estimates are in reasonable agreement with each other. Without more detailed information it is not possible to assess their validity. However, the population of farmed deer in the UK is only very small at about 26,000 animals (BBSRC, 1997b), and the accuracy of this estimate does not have a significant influence on the total NH_3 emissions.

3.3. AMMONIA EMISSION ESTIMATES FOR MINERAL FERTILISERS

The application of mineral N fertilisers is well established as a source of agricultural NH_3 emissions (e.g. Sutton *et al.*, 1995; Pain *et al.*, 1998). Other direct NH_3 emissions, such as foliage emissions from growing, senescing or decomposing vegetation have been discussed by Sutton *et al.* (1995) and Nemitz (1998). These foliar emissions are difficult to separate from direct fertiliser emissions, because they are both a function of the N application rates to crops and grassland. Sutton *et al.* (1993a) estimated 0.4 kg N ha^{-1} for these additional sources. For the purpose of this thesis, they were assumed to be included with fertiliser emissions, as they are effectively a function of the N supply (through N fertiliser application).

Ammonia emission estimates for N fertilisers have generally been measured as a percentage loss factor of N for different fertiliser types. This figure can then be multiplied with the average fertiliser N application rate for a crop (amount of fertiliser applied per unit area) to obtain total NH_3 emissions on an area basis. For instance, assuming a 2% volatilisation rate of NH_3 from a field with 200 kg N fertiliser applied per hectare, the NH_3 emission can be estimated at 4 kg NH_3-N ha^{-1} $year^{-1}$. While recognising that there may be additional emissions from grass cutting and decomposing vegetation (see Sutton *et al.*, 1997), these are not included here.

It has to be stressed again that the NH_3 loss from fertilisers is a function of several factors and average estimates have large variations associated with them (e.g. Van der Weerden and Jarvis, 1997; Whitehead and Raistrick, 1990). Some of the factors influencing NH_3 losses from fertilisers are the type of fertiliser (see Section 2.5.), soil properties (pH, water content, calcium content, buffer capacity, porosity), application timing and techniques as well as meteorological conditions during and after application (such as temperature, wind speed, precipitation). These as well as fertiliser requirements and recommended and actual application rates to crops and grassland have been discussed in Sections 2.2. and 2.5. Emissions from fertilised grazed grassland are included with grazing emissions from agricultural livestock (to avoid double-counting), and have been discussed elsewhere (Sections 2.4. and 3.2). This section discusses emissions from fertiliser application to crops and cut grass (for hay and silage), comparing emission estimates for different fertiliser types from recent studies (see Table 3.7.).

All studies agree that the highest volatilisation losses from fertilisers applied to crops and grassland are from urea. This is because the hydrolysis of fertiliser urea promotes NH_3 emissions in a similar way to emissions from urea in livestock manures (Sutton *et al.*, 1995). Emission estimates in the most recent studies range from 5-23% (see Table 3.7.) for urea.

Some studies (Jarvis and Pain, 1990; Van der Weerden and Jarvis, 1997; BBSRC, 1997b) provide separate volatilisation losses for grassland and arable crops, especially for urea. This is because fertilisers applied to arable land are assumed to be incorporated into the soil, whereas on most grass swards fertiliser is left on the surface. The latter results in a larger proportion of NH_3 lost through volatilisation, according to Van der Weerden and Jarvis (1997) by a factor of 2. (An exception to this is the injection of anhydrous ammonia and N solutions into grass swards on suitable soils. This does, however, not occur frequently in the UK.) A second reason for distinguishing between emissions from fertiliser applications to grass and arable crops is the difference in fertiliser types applied: According to BBSRC (1997b), conserved grassland in the UK receives on average 1.8% of its N fertiliser applications as urea, whereas arable crops receive 7% as urea. This partially makes up for the larger volatilisation loss rates from conserved grassland.

Other fertiliser types are associated with much smaller emission estimates of 0.8-8%, with averages of 2-3% across all studies. The only exception to this is ammonium sulphate with average volatilisation rates of 8-10% (Table 3.7.).

Table 3.7. Ammonia emission estimates from fertiliser application to crops and grassland.

	Jarvis & Pain (1990)	Asman (1992b)	Eggleston (1992)	ECETOC (1994)	Sutton *et al.* (1995)	TFEI (1996)[d]	BBSRC (1997b) grass	BBSRC (1997b) arable	Bouwman *et al.* (1997)[f]	avg. of arable estimates	avg. of grass estimates
urea	5%[a],15%[b]	15%	10%	15%	10%	10%[e]	23.0%	11.5%	15%	11.4%	14.1%
ammonium nitrate	0%	2%	10%	2%	1%	2%	1.6%	0.8%	2%	2.5%	2.6%
ammonium sulphate	-	8%	-	10%	-	10%	-	-	8%	9.0%	9.0%
anhydrous ammonia	-	1%	-	-	-	4%	-	-	4%	2.0%	2.0%
N solutions	-	2.5%	-	-	-	8%	-	-	2.5%	4.3%	4.3%
other (mainly NPK)	0%	4%	1%	8%	2.5%	2%	1.6%	0.8%	4%	2.8%	2.9%
UK average	0.6%[c]	3.8%	-	3.5%	2.4%	-	2.0%	1.5%	-	2.4%	2.5%

[a] arable, [b] grassland, [c] calculated by Sutton *et al.* (1995), [d] group II: temperate and warm-temperate countries,[e] preliminary version for the new edition (1998) includes: Van der Weerden & Jarvis (1997): emissions greater by factor 2 for urea on grassland, [f] after Asman, 1992b (based on Whitehead & Raistrick 1990), except for urea and anhydrous ammonia, which were estimated by Bouwman *et al.* (1997). 3.4. Other sources of NH_3 emissions

3.4. AMMONIA EMISSION ESTIMATES FOR OTHER SOURCES

Ammonia sources other than agricultural livestock and fertiliser N application to crops and grassland contribute about 15% of the total NH_3 emissions in the UK (e.g. RGAR, 1997). The minor sources summarised here as 'non-agricultural' or 'other miscellaneous NH_3 sources' include humans and pets, horses, wild animals, biomass burning, sewage works and landspreading of sewage sludge, transport, industry etc. (see Table 3.8.).

Table 3.8. Comparison of selected literature estimates of UK non-agricultural NH_3 emissions (from Sutton *et al.*, 1998).

Source	Eggleston (1992)	Lee & Dollard (1994)	Sutton *et al.* (1995)	Sutton & Fowler (1998)	Sutton *et al.* (1998)
Base year	1990	1990	1990	1996	1996
Human breath, sweat, infants, smoking	14.0	11.5	2.5 (0.6-5.8)	2.5 (0.6-5.8)	1.2 (0.4-5.4)
Horses	4.3	-	5.8 (2.5-10.7)	5.8 (2.5-10.7)	7.5 (3.5-12.7)
Cats & dogs	15.6	19.8	7.2 (2.5-9.9)	7.2 (2.5-9.9)	5.3 (2.5-8.3)
Wild animals & seabirds	-	-	0.8 (0.2-1.6)	0.8 (0.2-3.3)	5.2 (1.9-11.4)
Biomass burning	-	-	1.6 (0.2-6.6)	1.6 (0.2-6.6)	1.6 (0.2-6.6)
Ecosystems (natural soils)	9.9	-	0	0	0
Sewage (works & landspreading)	3.3	11.5-15.6	10.3 (3.1-19)	15.2 (4-29.6)	15.7 (4.8-31.1)
Landfill sites	3.3	-	3.3 (1.6-6.6)	3.3 (1.6-6.6)	3.3 (1.6-6.6)
Agro-industry (fertiliser production)	12.4	7.4	1.3 (0.7-2.7)	1.3 (0.7-2.7)	3.3 (3.3-5.0)
Agro-industry (sugar beet processing)	-	-	-	-	0.9 (0.6-1.2)
Other industries	-	-	-	-	5.6 (5.6-8.4)
Transport	0.2	1.4	0.8 (0.4-1.6)	8.2 (4.1-12.4)	8.9 (3.3-14.5)
Domestic & industrial coal combustion	-	3.5	3.5 (1.6-6.6)	3.5 (1.6-6.6)	2.2 (1.1-4.7)
Waste incineration	-	0.6-0.9	0.7 (0.3-1.6)	0.7 (0.3-1.6)	0.9 (0.3-2.1)
Household products	-	-	-	-	1 (0.3-4.1)
Non-agricultural fertiliser use	-	-	-	-	0.3 (0.02-2)
Total	63	56-60	38 (14-73)	50.2 (18.4-95.8)	62.9 (29.5-122)

Compared with previous studies, the latest estimates by Sutton *et al.* (1998) include some new sources such as household products, infants and cigarette smoking, which have been quantified for the UK for the first time. Some of the changes in the subtotals of the different miscellaneous sources are due to revised source activity information, others due to a re-assessment of emission source strength estimates with new literature. For instance, the amount of coal burnt in the UK has decreased significantly over the last few years (DoE, 1990; DTI, 1997) resulting in decreased emissions from this source. On the other hand, the number of vehicles fitted with catalytic converters has increased substantially, leading to higher emissions from transport (see Sutton *et al.*, 1998). Due to new restrictions on dumping sewage sludge at sea, the proportion being spread on land and thus the magnitude of NH_3 emissions from this source has also increased significantly. Emission source strength estimates for humans and pets were re-assessed with independent literature sources regarding N excretion rates. This led to decreases in the UK emission estimates for these sources.

While Sutton *et al.* (1998) provide the best current estimates of emissions from non-agricultural sources, the uncertainty estimates provided with these figures are still very large at ± 50%. More research is needed to improve the emission source strength estimates.

3.5. DISCUSSION AND CONCLUSIONS

The magnitude of NH_3 emissions from agriculture as well as from other miscellaneous sources varies considerably between recent UK inventories. This is mainly due to the differing estimates of NH_3 emission source strength applied to a given number of sources. Compared with other pollutants, large uncertainties remain unresolved regarding NH_3 emission source strength, although the most recent inventories agree more closely than the earlier studies.

This chapter has critically reviewed the most recent inventories for both agricultural and non-agricultural sources, focusing on emissions from agricultural livestock, which provide the largest contribution to the total NH_3 emitted in the UK. While there is good agreement on some aspects, it is suggested here that more research is required to establish more reliable source strength data. For instance, there are large differences in opinion regarding the amount of N excreted by sheep (6-18 kg ewe^{-1} $year^{-1}$). However, all the studies discussed agree that approximately 7-8% of the total N excreted by sheep are volatilised. The same problem is encountered for pigs, with annual N excretion rates ranging from 9-17 kg for average pigs. Emission source strength estimates from cattle, poultry and mineral fertiliser application to crops and conserved grassland are more similar for the recent studies reviewed here, but still show significant uncertainties in source strength estimates. Overall, the BBSRC studies (BBSRC, 1997a and 1997b) provide the lowest estimates for most categories. The estimates of TFEI (1996) were chosen here as best estimates from the selection of recent inventories summarised in this chapter. This is because they appear to be the most realistic figures available, when compared with basic literature and other recent inventories. Furthermore, the authors of the BBSRC studies pieced their estimates for the different livestock categories together from experimental data, without taking the flow of N through the different stages of manure management (see Section 3.2.1.) into account. This resulted in some imbalances in their estimates.

The component emission estimates shown in Tables 3.3.–3.6. are important input parameters for a spatially distributed emission inventory from agricultural livestock. Grazing, housing & manure storage and landspreading of manures are associated with different areas on an individual farm and also with different landuse/landcover types over the country. This is especially important for large scale inventories on a field-by-field scale (see Chapter 8), as well as for large areas of the national inventory (see Chapters 4-7). For the purpose of spatially distributed emission inventories, housing and storage emissions can be treated together, as they are likely to occur in close proximity.

For practical reasons, only the component loss rates from the suggested best estimates (TFEI, 1996) were applied in the NH_3 source distribution model developed for this thesis (see Chapters 4 and 5). The overall source strength estimates for the different livestock categories and mineral fertiliser application were taken from DoE (1995), which provided the most up-to-date 'officially agreed' emission source strength estimates for the UK at the time the model development for this thesis was undertaken. The choice of the less than ideal source strength estimates derived from DoE (1995) for the main results section of this thesis was, however, determined by the need to bring the results in line with commitments to MAFF and EMEP. However, the effects of applying other studies' source strength data (BBSRC, 1997b; TFEI, 1996) on the spatially distributed inventory developed in this thesis were also explored, and the results of this sensitivity analysis are presented in Chapter 9.

For non-agricultural emission sources, the source strength estimates and source activity data from Sutton *et al.* (1998) were adopted as a best estimate. This is because this study provided the most recent and most in-depth investigation of the subject, which involved a review of the latest literature sources and activity statistics.

Chapter 4

Methodology for a national emissions inventory I: input data and implementation environment

4.1. INTRODUCTION

Mapping and modelling spatially distributed entities involves simplifying the real world by abstracting certain features and reproducing them in an understandable manner, while retaining their essential qualities (Figure 4.1.). These qualities depend on the modelling/mapping task at hand. Sometimes the spatial distribution of the entities can be measured and quantified directly from the real world, for other entities it has to be determined indirectly, due to lack of measured data. This lack may be due to a restriction in resources for gathering the data or due to there being no appropriate measurement techniques. Either way, the spatial distribution of such entities is inferred through the modelling process. Most spatially distributed environmental models depend to some degree on inferred as well as measured data. An example for this is the modelling and mapping of NH_3 emissions to produce a spatially distributed inventory at a suitable resolution.

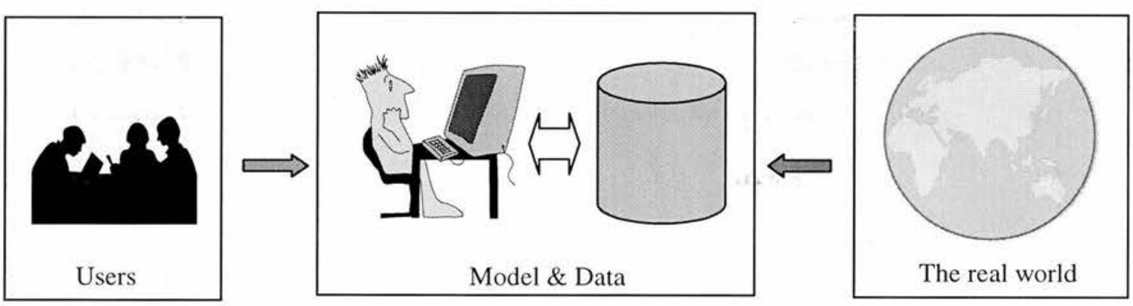


Figure 4.1. A schematic view of environmental modelling.

Earlier efforts to model the spatial distribution of agricultural NH_3 emissions by Kruse (1986), Eager (1992), Sutton *et al.* (1995) and Dragosits *et al.* (1996b) used a simple methodology, which was modified and substantially refined in this study. The different approaches are described and compared in detail in Chapter 5. Chapter 4 introduces the datasets used in the new approach presented here and discusses the choice of a suitable implementation environment.

4.2. DATA SETS

4.2.1. Agricultural Census

Agricultural livestock and fertiliser use on agricultural crops and grassland are the main source of NH_3 emissions in the UK. Quantitative and locational information on agriculture is therefore the single most important data source for the derivation of a spatially distributed NH_3 emissions inventory.

Data on agricultural and horticultural statistics for the UK are collected through an annual agricultural census. This complete survey of about 235,000 (GSS, 1996) 'main holdings' is conducted each June by the Ministry of Agriculture, Fisheries and Food (MAFF), the Welsh Office, The Scottish Office Agriculture, Environment and Fisheries Department (SOAEFD) and the Department of Agriculture for Northern Ireland (DANI). The definition of a holding can be roughly equated with the term 'farm'. The guideline definition is *"pragmatic and operational and subject to agreement with the individual farmer. The holding comprises land on which agricultural activities are carried out and which is by and large farmed in one unit having regard to such supplies as machinery, livestock, feedingstuffs and manpower, and to the distance between any separate areas of land involved and their type of farming. 'Farming' includes horticultural activity."* (GSS, 1996).

In addition, smaller holdings are surveyed periodically in England and Wales, most recently in 1994. In Scotland and Northern Ireland, a third of these 'minor holdings' is surveyed each year on a rotational basis. In total, there are approximately 77,000 minor holdings in the UK (GSS, 1996). The threshold which defines the difference between main and minor holdings has changed several times over the last decades (and varies for the different countries), which makes exact comparisons between certain years difficult (A.A. Bayley, Edinburgh University Data Library, pers. comm., 1996). This issue is discussed further in Section 7.3., where temporal changes between 1969 and 1988 are analysed.

At present, the minor holdings contribute about 25% of the total number of holdings in the UK (20% in England and Wales, 36% in Scotland, 34% in Northern Ireland). They represent, however, only a very small contribution to livestock numbers and crop areas (0.2-2.3% for different census items in England and Wales; GSS, 1996).

In Scotland, the minor holdings are spatially clustered, especially in the crofting areas of north-western Scotland, where they represent 56% of all holdings (SOAEFD, 1997).

Farmers are legally obliged to complete the census forms, answering approximately 200 questions relating to their agricultural activity, including numbers of livestock, areas of different landcover belonging to the farm (including woodland, rough grazing etc.) and areas of agricultural and horticultural crops including grassland on their farms (see Appendix A: copies of census forms). In order to ensure compliance, the individual holdings' census returns are made strictly in confidence, and are not disclosed to the public without aggregation to a suitable level to preserve this confidentiality. For this purpose, the census returns are currently aggregated to civil parish level or, if necessary, to larger areas such as parish groups.

The annual aggregated parish summary data are a unique dataset of considerable value for studies related to every aspect of agriculture, landuse, history and the environment. However, they also have limitations, which may cause more or less severe problems, depending on the nature of the investigation.

Coppock (1976a) discusses the representativeness of a data set collected in June for a whole year. He states that for most crops, with the exception of some vegetables, the June census estimates are probably the most accurate ones, because most crops are in the ground at the time of the census. A snapshot at the beginning of June is most likely to underestimate vegetables, since some areas are double-cropped, while others are harvested before the census date, or have not been planted yet by that date. Catch cropping may also escape enumeration. Regarding livestock, early June is a less representative date for a census than for crops, because of the mobility of livestock and the large seasonal movements that take place both locally and over long distances (Coppock, 1976b; see Figure 4.2.). This can lead to underestimations in some areas, with overestimations in others. For instance, beef cattle may be bred on one farm, reared on another and fattened on a third (Coppock, 1976a).

Another source of spatial uncertainty of census returns is the allocation of holdings data to civil parishes, i.e. within one civil parish's boundary. There are no maps of the spatial distribution of agricultural land allocated to the census holdings nor do the census returns detail which land parcels are associated with which agricultural activity. For census purposes, any holding is therefore allocated to the civil parish it is recorded in (There are about 900 civil parishes in Scotland and about 12,000 in England and Wales.).

However, the boundaries of a holding change over time through farm amalgamations, acquisition of land etc., and a holding's land may be located only partially inside the boundaries of the civil parish in which it is counted, with the rest in, most likely, the neighbouring parish(es) (see Figure 4.3.). This may lead to significant over- or underestimation of livestock numbers as well as crop areas for some parishes.

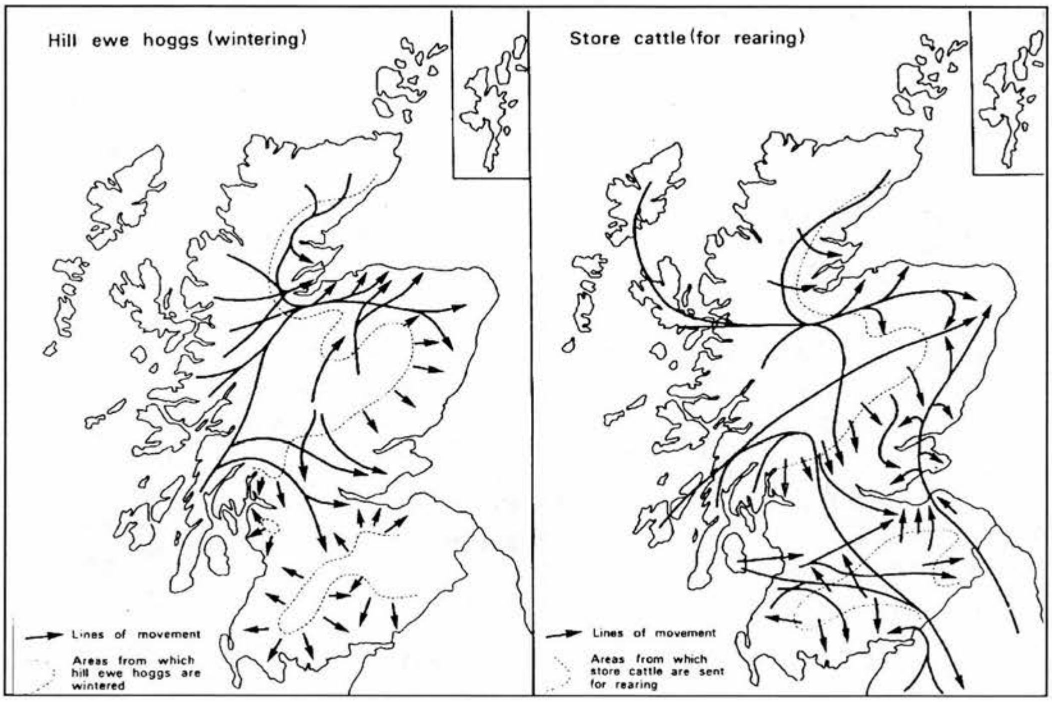


Figure 4.2. Livestock movements for sheep wintering and cattle rearing in Scotland, 1970 (from Coppock, 1976b).

Coppock (1965, 1976b) suggests that for the majority of land this will not cause great problems, although it may explain some apparent anomalies for certain parishes. This is an element of uncertainty in the census that can not easily be resolved without direct access to the original holdings data.

For this study, parish census data (and the corresponding parish boundaries at a 1 km by 1 km grid resolution) from all main holdings were available for 1988 for Great Britain. For 1996, the census data from all main holdings in Great Britain were provided at holdings level. Furthermore, the 1994 data for minor holdings for England and Wales were also available. In total, this amounted to approximately 250,000 holdings, with up to 162 data entries from livestock and crop statistics per record (162 for main holdings in England and Wales, 93 for main holdings in Scotland, 39 for minor holdings in England and Wales). These detailed holdings data had to be summarised to parish level, due to the confidentiality and disclosivity agreement with MAFF and SOAEFD.

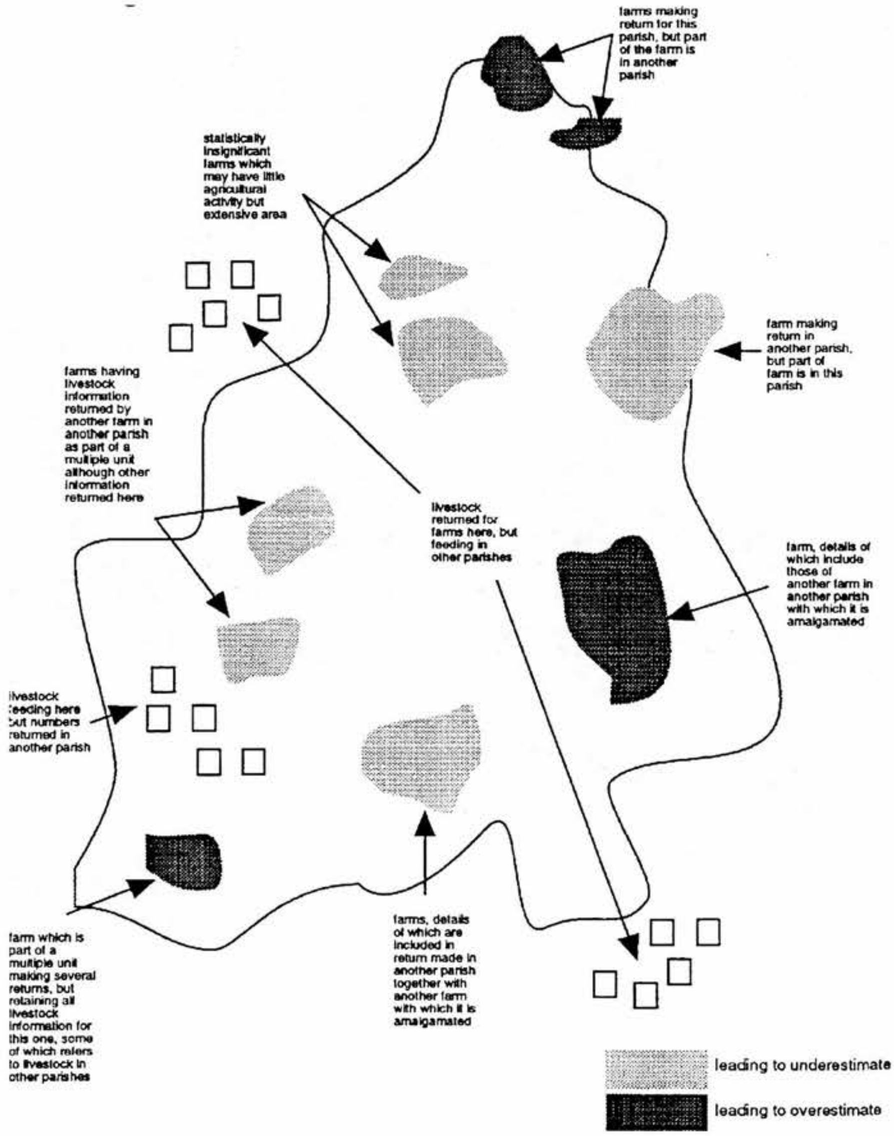


Figure 4.3. Limitations of Parish Agricultural Statistics (from Hotson, 1988).

For Northern Ireland, data were provided for 1996 only. The agricultural census data were supplied on a 5 km by 5 km gridsquare basis, summarised from holdings data by DANI.

In England and Wales, where the civil parishes are much smaller than in Scotland, it is much more likely that there are insufficient holdings in any one parish to pass the threshold set by MAFF to ensure non-disclosivity of data for an individual holding. (For example, the average parish size is 13.6 km^2 in England & Wales, with a range of 1-410 km^2, while parishes in Scotland have an average size of 95.0 km^2, with a range of 6-1124 km^2.) This resulted in the creation of combined parishes for England & Wales to provide the non-disclosive parish dataset supplied for 1988 by the Edinburgh University Data Library for this thesis: the census data for all thus affected parishes in each county were amalgamated into a spatially discontinuous county summary (SDCS) parish.

The major disadvantage of this concept is obvious when the data are used for spatial modelling: significant numbers of livestock or crops may be displaced from their original location over a whole county. Regarding the spatial modelling of NH_3 emissions, this may result in artificial 'hot spots' or disguise real problem areas entirely from the resulting maps. This issue is explored further in Sections 7.2. and 9.2.1.

For 1996, it was possible to negotiate access to the original disclosive holdings data by parish, which were then summarised to parish level according to disclosivity rules agreed with MAFF. The condition for use of the disclosive data was that the model output resulted in non-disclosive NH_3 emission maps, i.e. did not represent information on less than 5 holdings contributing to any output unit in the result. This required the amalgamation of all parish data which would result in disclosive output rather than the amalgamation of all disclosive input to the model. This was accomplished in a different way from the SDCS parishes used by MAFF. Potentially disclosive parishes were amalgamated with neighbouring parishes, rather than all other potentially disclosive parishes in each county. Thus, the spatial dislocation of NH_3 sources was kept to a minimum. The potential effects of a SDCS parish approach for 1996 are analysed and discussed in Section 9.2.1.

There are small differences in the census questionnaires between the different countries, especially for the earlier data, as well as between the years. Recently efforts have been made to make the censuses more comparable. For the purpose of NH_3 emission estimation, however, these problems could be resolved through the aggregation of livestock and crop subclasses. Furthermore, some census items are removed from or added to the questionnaires over the years, as certain crops are not worth the effort of recording anymore or new crops or animals are successfully introduced to UK agriculture. Examples of this are the loss of differentiation for orchard fruit or glasshouse fruit and vegetables from the census as well as the introduction of farmed deer or oilseed rape into the census.

The agricultural census data for 1970 were mapped and analysed at parish level for Scotland and at district level for England & Wales by Coppock (1976a, 1976b) to provide an agricultural atlas for Great Britain. A more refined method was developed at the Edinburgh University Data Library (Hotson, 1988) to map the parish summary data in grid format. These data have been derived for most years since 1969 at various resolutions from a 1 km to 100 km gridsize, although there are concerns about the accuracy at the 1 km resolution (see Chapter 5). Building on the work at the Data Library, this study developed a new methodology, which redistributes the census data for the specific purpose of modelling the spatial distribution of NH_3 emissions. Both the more general approach developed by Hotson and the new methodology are described in detail in Chapter 5.

4.2.2. Landuse/landcover data

Agricultural landuse data are one of the main information sources to link NH_3 emissions to specific locations in the landscape, or, more specifically, within each parish. As Hotson (1988) indicates, the areal definition of the agricultural census data can be improved without violating the disclosivity rules imposed. This can be achieved by spatially redistributing the parish census data with the aid of landuse data.

Landuse, however, cannot easily be observed directly (Wyatt *et al.*, 1990), although it may be inferred from observations of landcover. Any country wide landcover map will have associated errors and uncertainties which are unavoidable, and reasons for these can be found in the data collection and analysis methods (see Section 9.2.2. for a fuller assessment of this dataset). Surveyors may misclassify what they see on the ground, features on aerial photographs are not always unequivocally interpretable, or satellite image classifications will always contain a certain amount of misclassified areas. Similar features are more likely to be confused with each other than dissimilar ones.

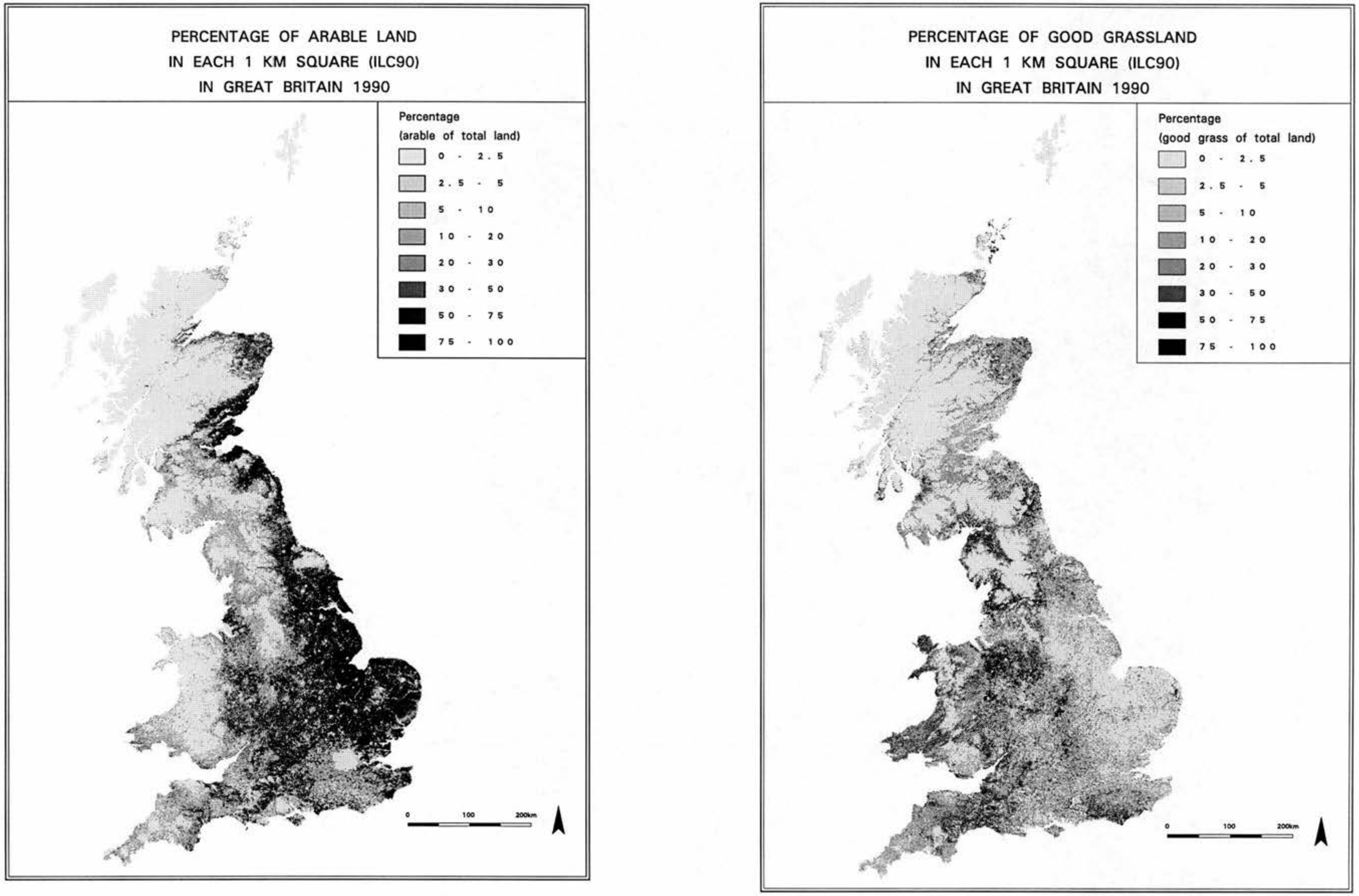
For this study, a number of available landcover maps were considered. These are the ITE satellite landcover map (in the following referred to as ILC90) (see Fuller *et al.* 1994, Wyatt *et al.*, 1990), the Countryside Survey 1990 (CS90) (Wyatt *et al.* 1990, Barr *et al.* 1993), the CORINE landcover map (O'Donovan *et al.*, 1993), the landcover dataset derived at the Edinburgh University Data Library by Hotson (1988) and the Landcover of Scotland 1988 (LCS88) map (MLURI, 1993). All these datasets were considered of sufficient quality for the purpose of this study, regarding the uncertainties discussed above, and the final selection was made on the grounds of areal coverage, spatial resolution and availability of the datasets.

One of the aims of this study was to avoid discontinuities at the border between different countries if at all possible. For this reason the LCS88 dataset was rejected as it is limited to Scotland, and no equivalent dataset for England and Wales exists. The CORINE landcover map was not used for the same reasons, as it has so far only been published for Ireland. Although this dataset could have been used for Northern Ireland, it was not used due to the format of the agricultural census data available for Northern Ireland.

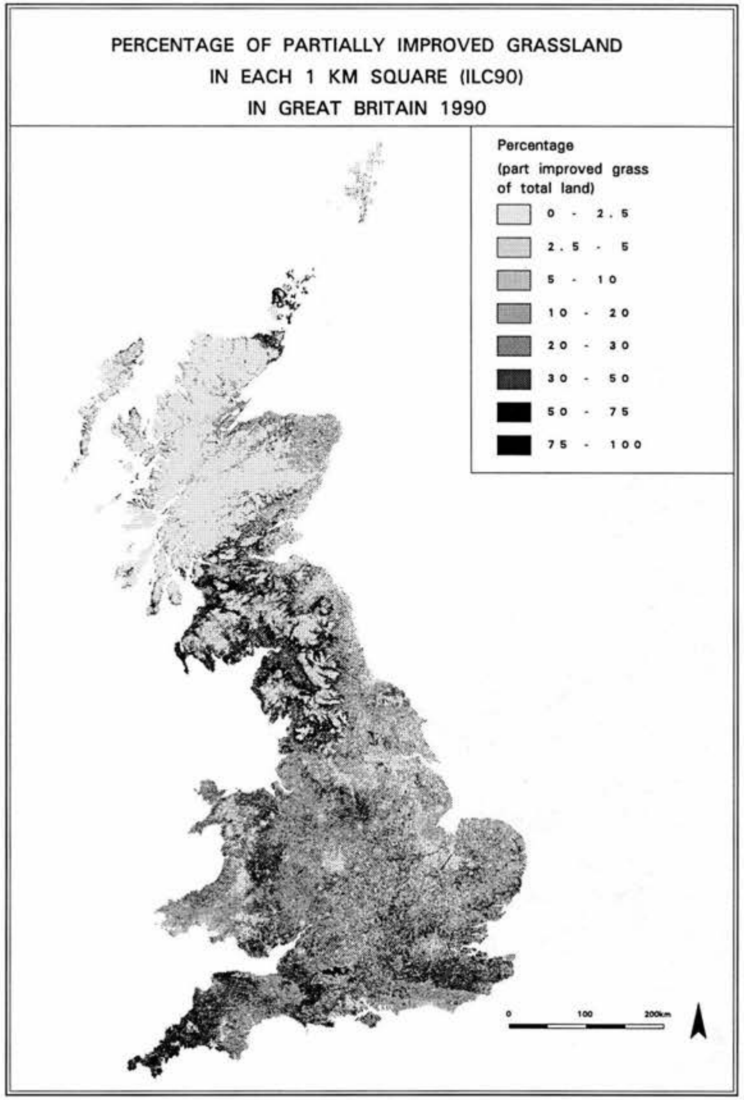
Hotson's (1988) dataset was specifically designed and used for the spatial redistribution of parish census data onto a regular grid. Compared with the ILC90 dataset it has, however, several disadvantages. Hotson's 1 km grid map dates back to the late 1970s, and each square kilometre is assigned to one of three classes: core agricultural land, moorland and land excluded from agricultural use (urban areas, inland water, forest etc.). The ILC90 dataset, on the other hand, provides more recent information (base year 1990 $\pm$ 2 years). It was derived from classified multi-season satellite (LANDSAT Thematic Mapper) images of Great Britain at a 25 m resolution (Fuller *et al.*, 1994). For each 1 km grid cell, the original 25 m pixel values were classified (26 landcover types) and aggregated to percentage values for each landcover type within the 1 km cells. These cover types were aggregated to selected 6 classes for the purpose of this study: tilled land, good grassland, partially improved grassland, poor rough grassland, very poor rough grazing land (heather etc.) and suburban/rural development, as shown in Table 4.1. Compared with Hotson's landcover map, the ILC90 data are believed to provide a resolution better suited to NH_3 emission modelling, because they do not exclude agricultural activity from any one square by definition, but instead provide a proportional figure of agricultural potential. They also allow a more refined modelling of NH_3 emissions from different sources, because of the greater detail in the classification. The ILC90 dataset was therefore chosen for this study in preference to the other datasets, due to it being based on the most recent information, its spatial resolution and its detailed landcover classification.

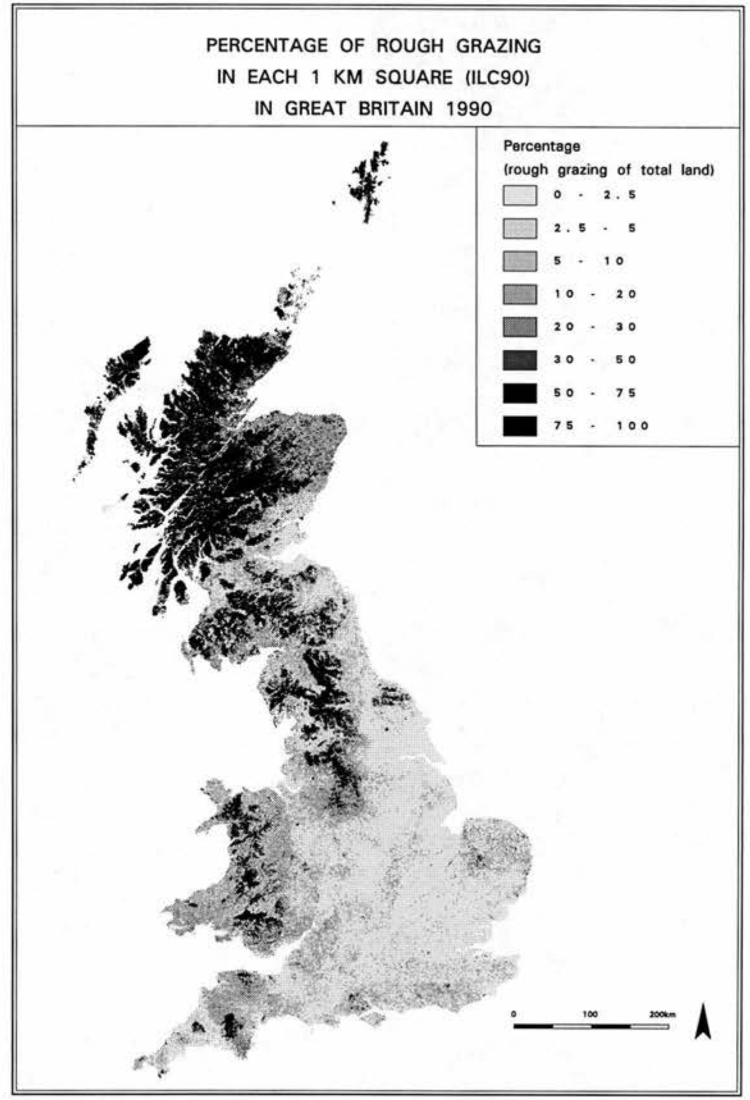
Table 4.1. Land cover categories of the Landcover map of Great Britain (ILC90), from Barr *et al.*(1993), and aggregated classes for use in the NH_3 source distribution model.

Landcover Map of Great Britain (25 categories)	Aggregated categories for this study
Sea/estuary	-
Inland water	-
Beach and coastal bare	-
Saltmarsh	-
Grass heath	Poor rough grassland
Moorland grass	Poor rough grassland
Mown/grazed turf	Good grassland
Meadow/verge/semi-natural	Partially improved grassland
Ruderal weed	-
Felled forest	-
Rough/marsh grass	Poor rough grassland
Open shrub heath	Poor rough grassland
Open shrub moor	Poor rough grassland
Dense shrub heath	Very poor rough grazing
Dense shrub moor	Very poor rough grazing
Bracken	-
Scrub/orchard	-
Deciduous woodland	-
Coniferous woodland	-
Lowland bog	-
Upland bog	-
Tilled land	Tilled land
Suburban/rural development	Suburban/rural development
Continuous urban	-
Inland bare ground	-
Unclassified	Unclassified

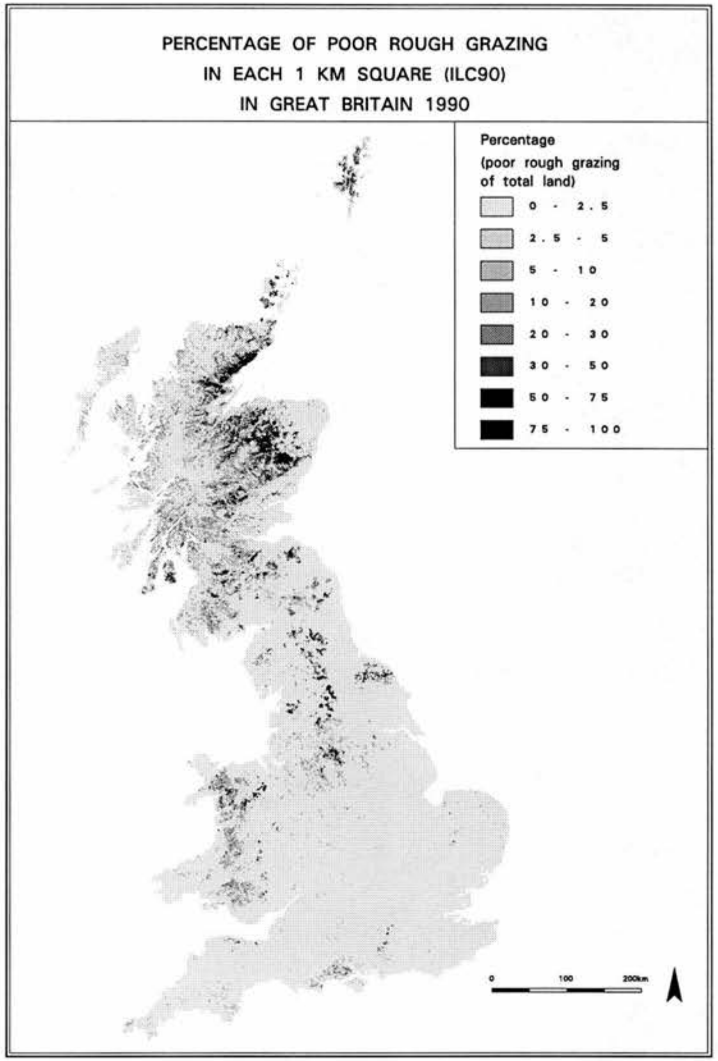


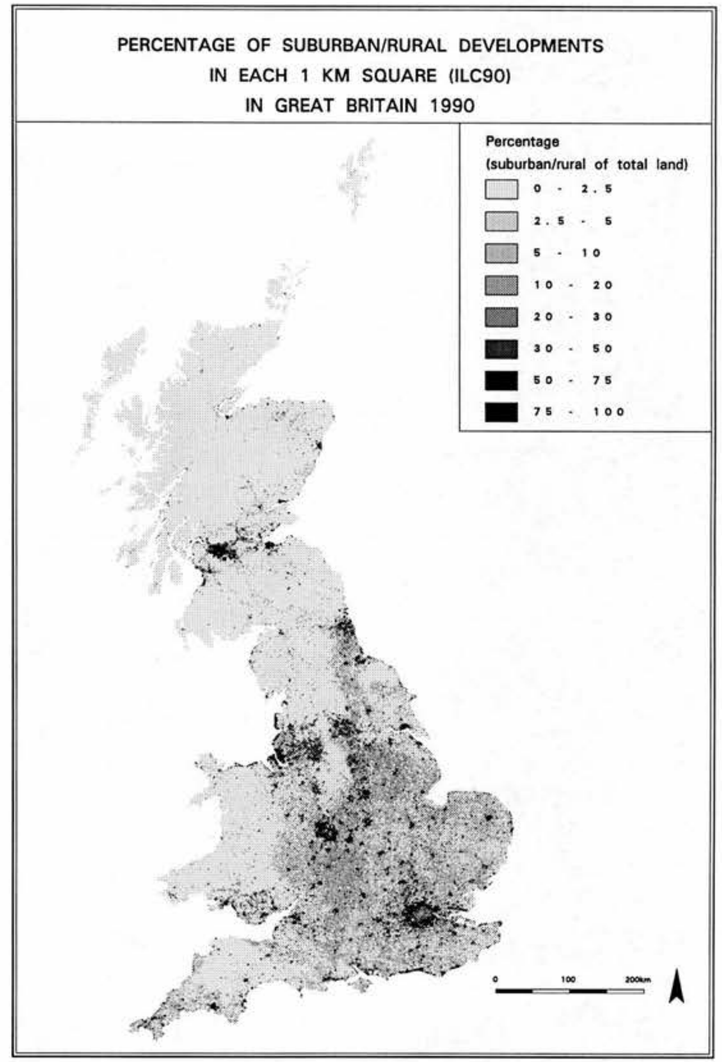
Figures 4.4.a-f: Landcover classes aggregated from the ILC90 dataset for use in the spatial distribution model for NH_3 sources: a) tilled land, b) good grassland.





Figures 4.4.a-f: Landcover classes aggregated from the ILC90 dataset for use in the spatial distribution model for NH_3 sources: c) part improved grassland, d) rough grazing.





Figures 4.4.a-f: Landcover classes aggregated from the ILC90 dataset for use in the spatial distribution model for ammonia sources: e) poor rough grazing, f) suburban/rural developmei

4.2.3. Emission source strength data

Ammonia emissions per unit livestock or per hectare crop vary and are dependent on many environmental factors and differences in agricultural practice between farms (see Chapters 2, 3). For instance, annual emissions from grazing livestock such as cattle or sheep vary greatly with the amount of N applied to pastures.

The greater the proportion of the year the animals spend outdoors grazing, the smaller the total annual emissions per animal are expected to be, due to lower loss rates during grazing (see Chapters 2, 3). This depends partially on the potential maximum length of the grazing season, which in turn depends on climatic and topographic factors, but also on the husbandry regime a farmer chooses to apply. Some of these factors could be modelled to a certain degree, provided sufficient spatial data and supporting emission source strength information are available (see Chapter 10). However for the present estimates, average conditions over the whole country were assumed.

Table 4.2. Ammonia emission estimates for agricultural sources in the UK 1996 after DoE (1995), Sutton *et al.* (1995) and TFEI (1996). Totals may not add up to the last decimal place due to rounding.

Category	Animal numbers (UK)	Emission/animal (kg NH_3-N $year^{-1}$)	Total NH_3-N (kt $year^{-1}$)	Contribution (%)
Cattle	11,904,000	11.23	133.7	57.4
Sheep, goats	41,623,000	0.38	15.9	6.8
Pigs	7,506,000	3.18	23.9	10.3
Poultry	146,496,500	0.19	27.8	12.0
Horses	302,000	6.56	2.0	0.9
Deer	33,700	0.95	0.03	0.01
Total livestock	-	-	203.2	87.3
Crops & grassland	-	volatilisation: 2.94% (of N fertiliser applied)	29.7	12.7
Total emissions	-	-	232.9	100.0

Total annual NH_3 emissions for each livestock type were derived from the official NH_3 emission figures (DoE, 1995), as well as from Sutton *et al.* (1995) and TFEI (1996) for farmed deer and horses (see Table 4.2.). The 'officially agreed' emission figures were chosen for this study in the first instance, amended for deer and horses, to illustrate the new spatial distribution methodology developed and to discuss the basic results. In Section 9.3. other inventories' emission source strength estimates (BBSRC, 1997b; TFEI, 1996) are used as input data to the same source distribution and emissions model to test the model sensitivity and assess the influence of uncertainties related to emission source strength estimates on the results of the model.

In detail, NH_3 emissions from agricultural livestock may be assigned to four main components:

- livestock housing
- manure storage
- landspreading of livestock manures
- livestock grazing

Ammonia emissions from each livestock type under average husbandry conditions can be apportioned to these different NH_3 emission components (Table 4.3.; see also Chapter 3). For the purpose of the spatial distribution methodology developed in this thesis, livestock housing and manure storage emissions were treated together as they occur in close spatial proximity. The estimates derived from DoE (1995) do not provide details on the relative proportions of NH_3 emitted during the different livestock husbandry stages, and these proportional values had to be adopted from another source. For this purpose the data of TFEI (1996) and Sutton *et al.* (1995; for cattle) were chosen, because they provided the best compromise between all the different estimates presented in Chapter 3. For farmed deer, the same proportions as for sheep and goats were assumed.

Table 4.3. Proportions of NH_3 emission components for livestock classes, derived from TFEI (1996) and Sutton *et al.* (1995; for cattle); (individual percentages may not add up entirely due to rounding).

%	Dairy cows	Other cattle	Fattening pigs	Sows	Sheep	Horses	Laying hens	Broilers	Other poultry
Housing + storage	45	45	58	58	18	37	62	60	60
Spreading	37	37	41	42	16	27	39	41	41
Grazing	18	18	0	0	65	37	0	0	0

Ammonia emissions from the application of mineral fertiliser to crops and conserved grassland are mainly dependent on the fertiliser type and N fertiliser application rate, as discussed in Chapters 2 and 3. The N fertiliser application rates typical for crops in Great Britain were available in detail for the main crops and crop groups from the British Survey of Fertiliser Practice (BSFP) for 1988 and 1996 (Chalmers *et al.*, 1989; Burnhill *et al.*, 1997). For Northern Ireland, the BSFP values for Great Britain were applied, in the absence of specific application rate estimates for Northern Ireland. The application rates are summarised in Table 5.5. (Chapter 5). There is potential for deriving spatially variable fertiliser application rates through spatial interpolation methods from the BSFP sample data. This provides an insight into one of the main uncertainties in the spatial distribution of estimated NH_3 emissions from crops and cut grassland and is discussed in detail in Chapter 10.

The fertiliser type (see Sections 2.5, 3.3.) was simplified and assumed to be evenly distributed over the country and all crops. An estimated average volatilisation factor of 2.94% of the applied mineral N fertiliser was derived from the official emission figures (Table 4.2.; DoE, 1995, Dragosits *et al.,* 1996b).

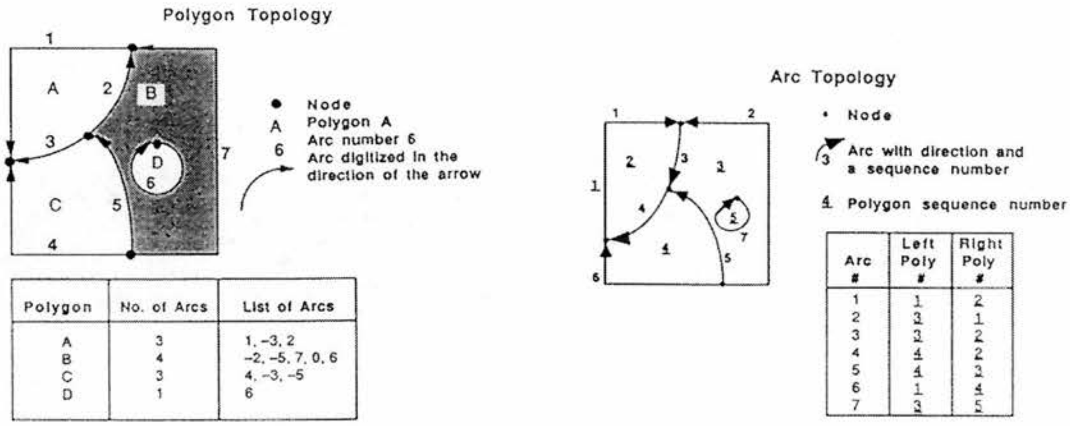
For other miscellaneous NH_3 sources such as humans, pets, industry, transport etc. (see Section 3.4., Table 3.9.), the estimates of Sutton *et al.* (1998) were chosen. The methodology for spatially redistributing these sources is described in Section 5.6.

4.3. IMPLEMENTATION ENVIRONMENT

Environmental modelling with large volumes of spatial data is a complex undertaking. It is therefore essential to ensure the model is developed in the most efficient way and the data are represented in the best possible format for the purpose of the model. This involves deciding on a suitable implementation environment for the model as a first step, including the spatial representation of the model input data, the model structure and the tools for developing the model and analysing the results.

The spatial location of entities plays a central role in the processing of geographical data. These entities need to be described by their location and also by any unique characteristics essential to the modelling process. The spatial information is usually described with the help of a co-ordinate system, in this case the UK Ordnance Survey National Grid. In general, there are two main methods of describing entities for the purpose of mapping and analysing spatially referenced data: the vector and the raster/grid based data models.

The vector model generalises entities into points, lines and areas. Additional information is attached as attribute data to each entity via unique identifiers. In addition to the geometrical information, topological information for each entity can also be derived, i.e. information on spatial relationships between the entities such as neighbourhood, hierarchy or connectivity (see Figure 4.5.).

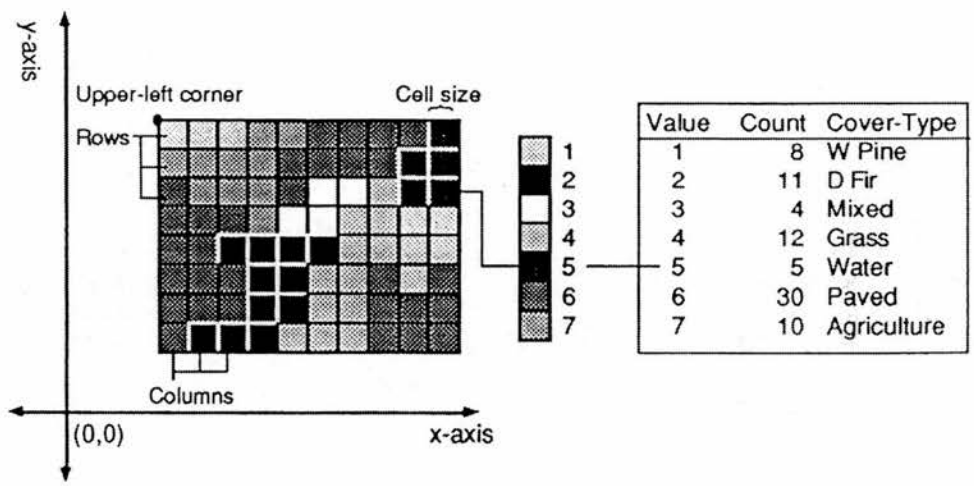


Polygon	No. of Arcs	List of Arcs
A	3	1, -3, 2
B	4	-2, -5, 7, 0, 6
C	3	4, -3, -5
D	1	6

Arc #	Left Poly #	Right Poly #
1	1	2
2	3	1
3	3	2
4	4	2
5	4	3
6	1	4
7	3	5

Figure 4.5. The vector model (from ESRI, 1991c).

The raster model (see Figure 4.6.) divides the study area in small (most frequently) rectangular cells. The spatial information is in general stored implicitly, i.e. only in relation to the origin of the co-ordination system.



Value	Count	Cover-Type
1	8	W Pine
2	11	D Fir
3	4	Mixed
4	12	Grass
5	5	Water
6	30	Paved
7	10	Agriculture

Figure 4.6. The raster model (from ESRI, 1991a).

The geometrical and topological information inherent in spatial data allows vertical and horizontal integration of the data. Vertical integration through 'overlay' of several datasets (e.g. landcover, parish census data) allows modelling and analysis through the combination of several datasets (see Figure 4.7.). Horizontal integration of spatial data through their topological relationships allows the modelling and analysis based on neighbourhood and/or distance concepts.

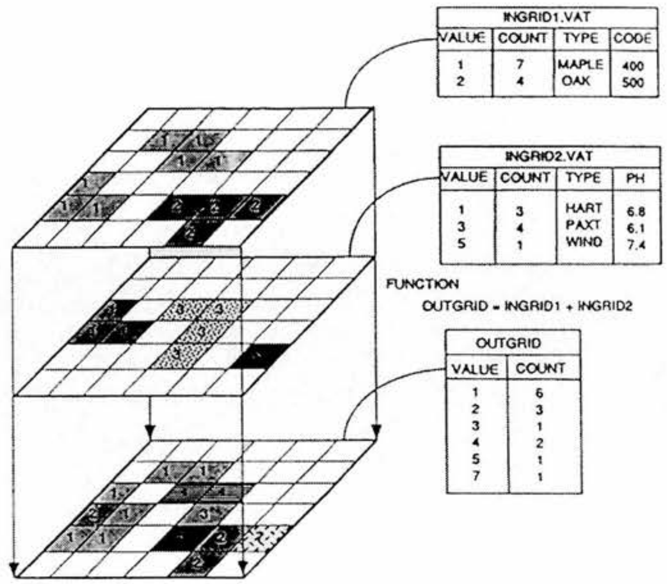


Figure 4.7. Vertical integration of spatial data (from ESRI, 1991b).

For this study, the raster representation was chosen for several reasons. Specifically programmed models, which involve vertical and horizontal integration of the input data, are much more efficient and faster in a matrix environment, as well as easier to code. In addition, most of the input data required for this study, landcover data and civil parish boundaries, were readily available in raster format. Furthermore, atmospheric transport and deposition models require spatially distributed NH_3 emissions in grid format.

Large spatially distributed datasets are advantageously managed in Geographical Information Systems (GIS). *"A Geographical Information System is a system for capturing, storing, checking, integrating, manipulating, analysing and displaying data which are spatially referenced to the earth. This is normally considered to involve a spatially referenced computer data base and appropriate application software"* (DoE, 1987).

All these capabilities and the specific functions available for horizontal and vertical integration appear to make GIS an ideal tool for the implementation of environmental models in general and a spatial emissions inventory in this study. There are, however, other considerations to take into account: the more complex the modelling task, the less efficient most GIS software products in general (and their macro languages in particular) are for the modelling of the processes themselves (see

Dragosits *et al.*, 1996a; Reyes *et al.*, 1993). For the NH_3 source distribution model developed for this thesis, several spatial datasets with approx. 230,000 grid cells are integrated with the parish census data, totalling over 4 million calculations, each of which involves several conditional equations.

By coupling complementary systems such as GIS and (external) environmental models, the resulting combination, also known as 'hybrid systems', should be more efficient (Nyerges, 1992). For these reasons, it was decided for this project to develop a hybrid system, with the spatially distributed modelling performed outside the GIS. The storage of spatial model input data and results, mapping and display as well as further analysis were carried out by a GIS (ARC/INFO), while the model for the spatial redistribution of the NH_3 sources was written in FORTRAN77.

4.4. IMPORTANCE OF SCALE AND SPATIAL RESOLUTION

The scale and spatial resolution suitable for any model output are very much dependent on the model input and the assumptions and calculations inside the model. There are two main issues to consider: Firstly, the spatial resolution of the model output can only be as good as that of the input data. If the output data are presented at a much finer resolution than the input data and modelling process make feasible, the output may mislead users by its pseudo-accuracy, which should clearly be avoided. Secondly, uncertainties in the input data and the assumptions and rules of the model will be propagated and make the output data more uncertain than the input data at any given resolution. It appears therefore sensible to distinguish between a 'processing level', which matches the resolution of the input data to get the greatest level of detail, and a 'publication level'. The spatial resolution at the publication level will take account of the uncertainties involved at the processing level and the aim of the project.

For this study, the input data were available at a 1 km level (landcover and parish boundaries) and as parish summary data in table format. The uncertainties inherent in these data have been discussed above (Section 4.2.) and their influence on the uncertainties of the model output are considered in greater detail in Chapter 9. The main aim of this study was to produce a new improved spatially distributed NH_3

emissions inventory for the UK at the national level, with enough detail at the regional and even local level. To make the most of the resolution of the input data and get the best possible locational accuracy, the model was thus developed at a 1 km grid resolution. However, although the parish-based agricultural census data have a notional spatial resolution of 1 km, this is not strictly true: The real basic information unit of these data is the parish, which is of variable size and irregular shape. Thus, the output was aggregated to a 5 km grid for the following reasons:

1) The 1 km data are clearly much more uncertain than the more robust 5 km data. While the 1 km data should not be used to provide exact values at this resolution, they may in principle be useful, if treated with caution, as they give a statistical representation of the likely distribution at a finer resolution. They also provide an indication of the high spatial variability within the each 5 km output cell. In Chapter 5 (Figure 5.5.) model output data at the 1 and 5 km grid resolution are shown, highlighting both the benefits and limitations of the estimates at the 1 km level. Any use or publication of model output for 1996 at a finer than 5 km grid resolution, however, would need to be agreed with MAFF, due to potential disclosivity at the 1 km resolution.

2) The finest spatial resolution of current atmospheric transport and deposition models for the UK is the 5 km grid level at the national scale. These models and the study of the effects caused by atmospheric pollution are one of the main reasons for creating spatially distributed emission inventories, which are the main input data source for such models (see Chapter 1).

4.5. SUMMARY AND CONCLUSIONS

Environmental modelling of spatially distributed entities is a complex undertaking. Before embarking on the modelling process itself, general decisions regarding the approach best suited for the aims and objectives of the study have to be made. This involves considering available input data sources and the implications of using them, how to put them together in the model, etc. The main issues involved in this process are the choice of suitable data models and representations of the circumstances in the real world that the model is aimed to reproduce. Scale and resolution of the input data, the implementation environment as well as the way the processes are represented in the model have significant implications on the model output.

For this study, the main input data sources have been identified here:

a) the (parish-based) agricultural census,

b) the ITE satellite landcover data (ILC90),

c) the emission source strength data agreed on by a panel of scientists (DoE, 1995; TFEI, 1996; RGAR, 1997), and

d) the British Survey of Fertiliser Practice (BSFP).

Livestock emissions from different component sources, such as grazing, housing, manure storage and manure spreading, can be apportioned to the different landcover types where they occur. This provides the basis for developing a more detailed spatial distribution model of NH_3 sources.

With the large volume of data involved in the model proposed here for the spatial distribution of NH_3 sources in the UK, it is important to choose a suitable implementation environment. The most appropriate data model for the type of data and operations involved is a grid representation. This allows fast and efficient horizontal and vertical integration as well as providing the end product in a format suitable for immediate use in atmospheric transport and deposition models. A hybrid approach linking GIS and a purpose-built FORTRAN77 model at a 1 km grid resolution ('processing level') was chosen as the most efficient solution for a national NH_3 emissions inventory for the UK.

While the model output is aggregated to a 5 km grid dataset ('publication level') which provides more robust results than the 1 km data, the finer resolution output could in principle be useful for a statistical representation of the local variability of NH_3 emissions, if the related uncertainties are borne in mind. A publication of the model results for 1996 at a 1 km resolution is at present constrained, however, by the aim to preserve the confidentiality of the farmers, as agreed in a contract regarding disclosivity with MAFF and SOAEFD.

Chapter 5

Methodology for a national ammonia emissions inventory

II: Source distribution and emissions model

5.1. INTRODUCTION

The distribution of NH_3 emissions can be described as a function of the spatial pattern of source sector activities, i.e. chiefly agricultural livestock and fertilised crops and grassland. It is therefore important to recreate this spatial source activity distribution as realistically as possible in the model. The methodology described in this chapter consists of 2 main parts: a) the spatial distribution of agricultural source activities over the agricultural landscape and b) the assignment of NH_3 emission source strength data to the sources to calculate NH_3 emission maps. It is this latter part of the model on which other authors (Kruse, 1986; ApSimon *et al.*, 1987; Kruse *et al.* 1989; Eager, 1992; Sutton *et al.*, 1995; Dragosits *et al.*, 1996) have concentrated, taking the spatial distribution of agricultural livestock and crops from the existing general distributions of agricultural census data, e.g. by the Data Library (Hotson, 1988). This is a much more inflexible approach as it does not permit the inclusion of the characteristic spatial distribution patterns of NH_3 emissions, which are closely linked to the type and the intensity of the source activities in any particular location (see Section 5.5.).

The agricultural census data provide the main source of information regarding the spatial distribution of agricultural NH_3 sources. They are, however, aggregated to parish level for reasons of data protection (see Section 4.2.1.). Mapping these data in the simplest possible way would result in an even distribution of all recorded census items over the entire area of each parish, which is a very poor reproduction of real circumstances. By incorporating additional information such as landcover data, the spatial pattern of agriculture over the countryside can be modelled much more realistically. The purpose of the redistribution of the parish census data within the parishes' boundaries is to produce best estimates of the likely spatial distribution of agricultural activities at a given resolution, while maintaining confidentiality of the census returns. The logic behind this model is to allow for the absence of a census item for areas with unfavourable conditions, so that the total recorded is redistributed over the remaining area(s) of the parish. This approach was first developed in a simple model by Hotson (1988) in joint work with J.T. Coppock at the Data Library, University of Edinburgh (see Section 5.2. below). The results provide a reasonable picture of the general distribution of agricultural livestock and crops for Great Britain. The data resulting from this model were used at a 5 km grid resolution in previous national NH_3 emission inventories (Eager, 1992; Sutton *et al.*, 1995; Dragosits *et al.*, 1996b). These studies accepted the limitations posed by this coarser approach. They calculated their spatial inventories by assigning emission source strength estimates to the *census categories* provided and summing the results for each grid square.

For the purposes of a more accurate spatial NH_3 emissions inventory, it is, however, desirable to distinguish between intensively and extensively used agricultural areas, especially within the larger parishes. Emissions from livestock housing, storage and application of wastes are much larger and more spatially concentrated than from grazing animals. Thus, although animals may graze upland and hill areas at some time, most of the NH_3 emissions are located within better agricultural land at lower altitude (see also Chapter 2). Larger parishes, especially in upland areas such as Highlands of Scotland, tend to have large areas of very extensively used land. In this study, a new methodology was developed to specifically allow the redistribution of census data as *ammonia sources* rather than through a more general model, taking estimated emission source strength at the different stages of livestock management into account. This involved the apportioning of the average NH_3 emission potential of each agricultural entity (such as a dairy cow or pig) to the different NH_3 sources originating from it (housing and storage, landspreading, grazing emissions if applicable), as discussed in Section 3.2. and summarised in Table 4.3.

5.2. THE NEW METHODOLOGY FOR REDISTRIBUTING AGRICULTURAL AMMONIA SOURCES - GENERAL ISSUES

The main task of this study was to combine the available datasets to improve the spatial estimates of NH_3 emissions by incorporating other suitable information derived from landcover data and knowledge of agricultural practice, as there is not enough spatial detail in the parish summaries of the agricultural census. The new emissions model (Figure 5.1.) is based on the spatial redistribution of census data over suitable landcover types, using the same type of data and methods for all of Great Britain, so that discontinuities at the border between different countries are not an issue.

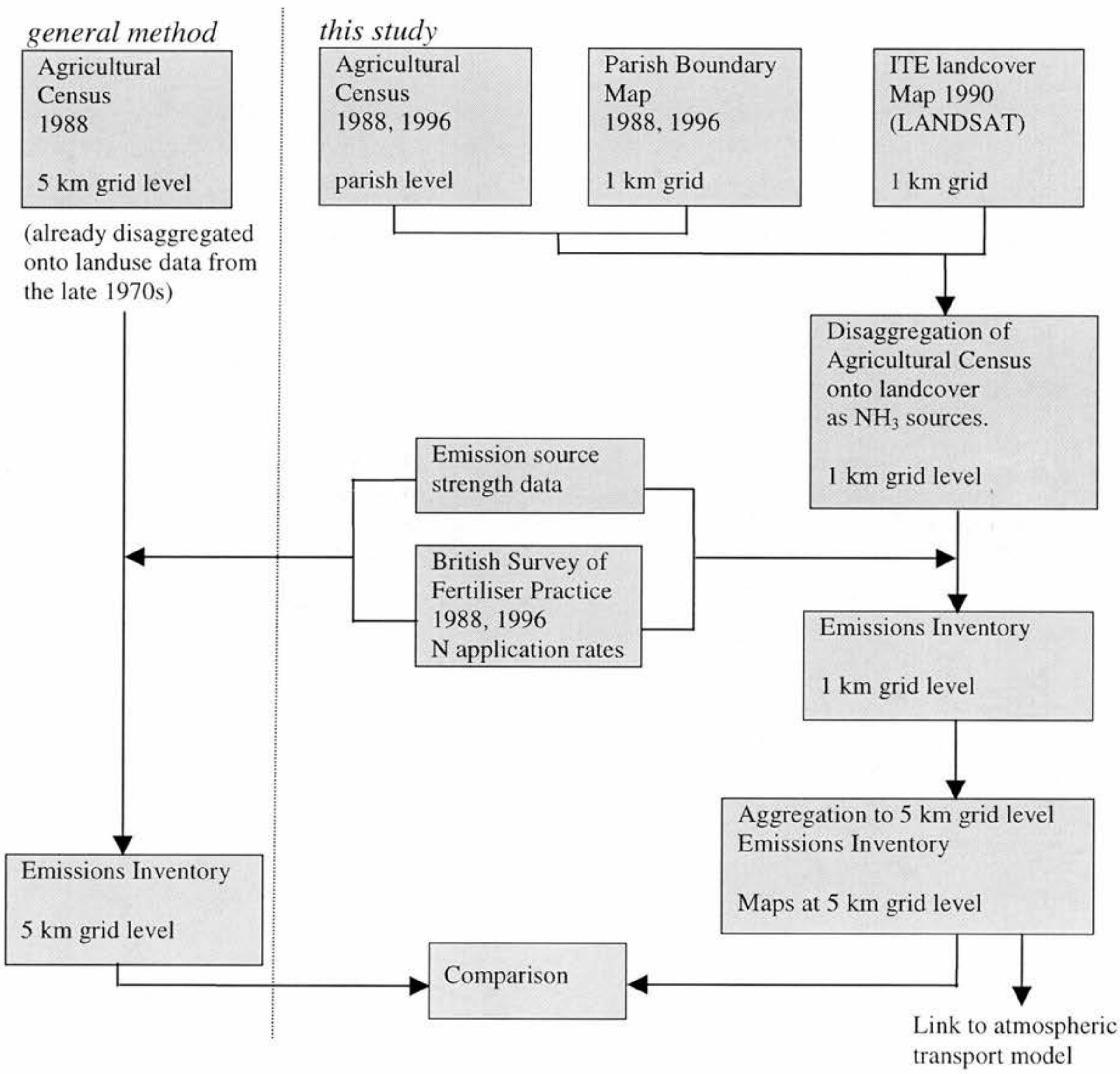
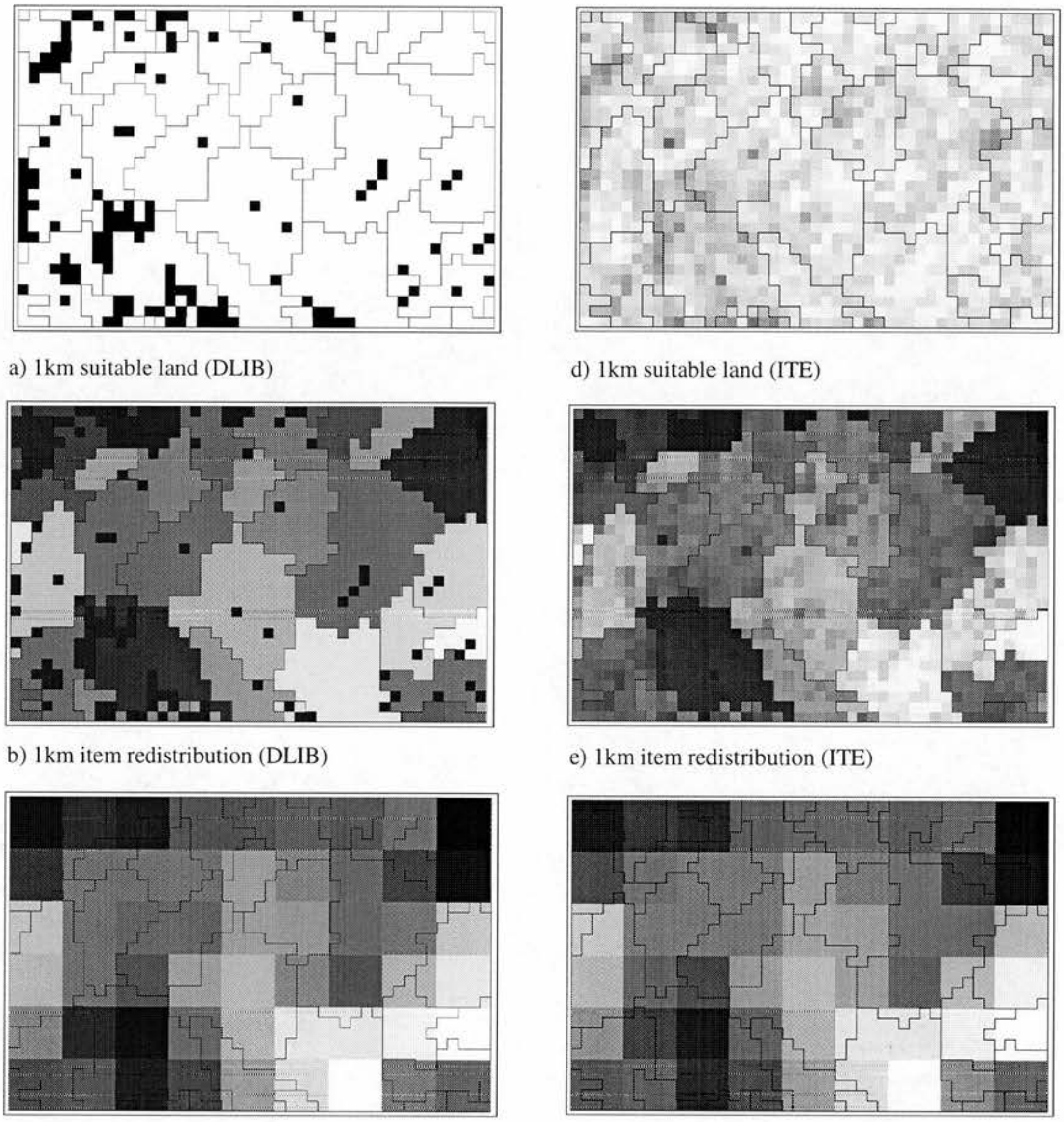


Figure 5.1. Methodologies for modelling the spatial distribution of NH_3 emissions (from Dragosits *et al.*, 1998)

Hotson's (1988) general method was modified in several ways: Firstly, individual census items were redistributed at a fine resolution (1 km grid) according to landcover and a weighting that is specifically related to the source strength of NH_3 emissions on different land-classes. Secondly, a more detailed landcover dataset was available which provided a percentage cover of different landcover types for each 1 km grid cell, rather than a single dominant class (see Section 4.2.2.). Furthermore, additional aspects were considered for future work, such as spatially variable NH_3 emission source strength estimates. An example for this is to allow for variations of the N fertiliser input to grassland in the model or the length of the grazing season and study its effects on livestock emissions over the country (see Chapter 10).

In a first step, the same parish census data were redistributed over the two different landcover datasets, applying the same basic allocation rules as stipulated by Hotson (1988), to check the effects of the 2 different landcover datasets and of applying the new methodology step by step (Figure 5.2.).

Figures 5.2.a and 5.2.d show that the ITE landcover data are more detailed and show the fraction of suitable land for distributing a particular census item. In this particular rural study area there are no absolutely black squares in the ILC90 data, since any 1 km square is rarely completely unsuitable. The resulting map of the census item distributed equally over the suitable land shows more structure between the 1 km grid cells in Figure 5.2.e than in Figure 5.2.b, and the boundaries between parishes with a similar density of this census item become blurred when the new landcover data are used in the model. On aggregation to the 5 km grid level (Figures 5.2.c and 5.2.f), however, the difference between the results using the two landcover datasets is relatively insignificant. This is because the rules for the spatial allocation of the census item for both models were deliberately chosen to be as similar as possible. The ILC90 data provide, however, the potential to redistribute items at a much more detailed level and to change the allocation rules specifically to suit the characteristics of NH_3 and the different source strength levels of different agricultural sources.



a) 1km suitable land (DLIB)

d) 1km suitable land (ITE)

b) 1km item redistribution (DLIB)

e) 1km item redistribution (ITE)

c) 5km level aggregation (DLIB)

f) 5km level aggregation (ITE)

Figure 5.2. Comparison of redistributing a census item (beef cows) in the Scottish Borders using the different landcover databases of Data Library (a-c) and ITE (d-e), but similar spatial redistribution rules. Black indicates absence and lighter shades increasing numbers.

From the above it should be evident that, although the primary intention is to provide maps at a 5 km grid resolution, it is necessary to first redistribute the census data at a finer scale. The 1 km grid scale of the parish boundary maps and landcover data provides an acceptable resolution for the main differences within a parish to be identified, as well as for the error in converting variable size parishes to a 5 km grid. It should be noted that the spatial distribution of NH_3 emissions, or any other comparable statistics, is ultimately limited by the spatial errors implicit in allocating data provided at a parish level (see Section 4.2.1.). The consequence is that, although

1 km grid resolution maps may be used as intermediate stages ("processing level"), they must be recognised as less reliable spatial estimates than at the 5 km resolution ("publication level").

If the landcover data were aggregated to a 5 km level before the spatial redistribution of the parish census data and the modelling performed entirely at a 5 km resolution, further uncertainties would be introduced to the results. This can be attributed to the fact that parishes have irregular shapes and sizes, which do not lend themselves to an aggregation to 5 km grid cells without the introduction of considerable spatial mis-location of the census data these parishes represent.

The choice of rules regarding the redistribution of agricultural census items in relation to landcover is very important in defining the overall distribution. In this respect, the rules for distributing NH_3 emissions are different to a general redistribution of census items. The main reason for this is that NH_3 emissions do not occur equally through all stages of livestock management, as summarised in Section 4.2.3. The emissions from these sources tend to occur on specific landcover types, i.e. livestock housing and manure storage will be located close to or on the holding, livestock grazing will mainly be taking place on grassland etc. Although the landcover types provided by the landcover map are not equivalent to land use (Wyatt *et al.,* 1990) (e.g. what the satellite classification identifies as grassland, could be a pasture or a football pitch), a strong correlation can be observed and landuse can be inferred. Thus the four main components of agricultural activities can be linked to different landcover types for the accuracy required of a national inventory at a 5 km grid resolution.

The methodology introduced above and described in detail below (Sections 5.3. and 5.4.) was applied to Great Britain only rather the whole United Kingdom. This is because the data available for Northern Ireland were at a different level of detail and a different spatial resolution (see Section 4.2.1.), and were used as provided by DANI in the emissions model (Section 5.5.).

5.3. THE SPATIAL DISTRIBUTION OF AGRICULTURAL LIVESTOCK AS AMMONIA SOURCES

It is especially important to set the parameters and rules guiding the spatial distribution of livestock emission sources in the model appropriately, as farm animals provide the largest contribution to UK NH_3 emissions. Agricultural livestock may be divided into two main categories: animals that are kept indoors throughout the year, and animals that are outdoors grazing for part of the year. The former category includes pigs and poultry, the latter cattle, sheep, goats and farmed deer. These categories are valid for average farming practice (see Chapter 2), with notable exceptions such as cattle kept indoors all year round or outdoor pigs and free range poultry. Without appropriate data regarding the spatial distribution of such practices, average conditions have to be assumed for the UK in the model. Some aspects of farming practice, however, such as the spatial variation of N fertiliser application rates to grassland or the average length of the grass growing season (and therefore the grazing season) can be approximated and studied in separate models (see Chapters 2 and 10). The results could in turn be incorporated into the basic NH_3 source distribution and emissions model described here.

In a first step, the parish livestock data and parish boundaries from the censuses for England & Wales and Scotland had to be amalgamated to a dataset for Great Britain. This involved revisions of some minor discontinuities regarding parish boundaries at the English-Scottish border and the amalgamation of parish data (and the corresponding boundaries) for approximately 50 parishes which would potentially lead to disclosive results (see Section 4.2.1.). During this process, the very detailed livestock categories were aggregated to a level suitable for NH_3 emission estimation, based on the availability of source strength data. This also entailed ironing out minor differences in the classification systems of the two censuses. The resulting livestock categories for Great Britain are listed below (Table 5.1.).

Table 5.1. Aggregation of Agricultural Census items for input to the spatial redistribution and emissions model.

Livestock categories (1988,1996)	Census item numbers (England & Wales)	Census item numbers (Scotland)
Dairy cows & heifers in milk & in calf	70,72	100,102
Beef cows & heifers in milk & in calf, bulls	71,73,74,76,78,80,81,94,95	101,103-105,108,110-113
Other cattle 1-2 years	75,77,79,83-86	106,107,109,114-117
Other cattle < 1 year	87-91	118-121
Other cattle (sum of 3 categories above)	71,73-91	101,103-121
Sows	100-102	146-148
Other pigs (fatteners, boars)	103-110	149-156
Laying & breeding hens	121,123,124,126,133,134	158-163
Broilers	127,128	164
Turkeys [a]	135	168,169
Other poultry (ducks, geese, etc.)	129,130,138	167
Sheep (> 1 year)	112-115	139-141
Lambs	116-118	143,144
Goats (all ages)	139,142,143	97,98
Farmed deer (all ages) [a]	96	94
Horses & ponies [a]	125,131	95,96

[a] Turkeys, farmed deer and horses & ponies were not included in the census questionnaire for England & Wales in 1988.

The model apportions these livestock categories derived from the census data to the best suited landcover classes within each parish. Each livestock category is distributed as an NH_3 emission source onto the appropriate landcover, by assigning % values to emission source strength estimates at the different livestock management stages (summarised in Table 5.2. below). The main objective here was to distribute emission sources to where they were most likely located on the ground.

Table 5.2. Emission sub-source distribution over different landcover types for livestock categories for input to the model (total fractions of NH_3 emissions from TFEI (1996) and Sutton *et al.* (1995; for cattle), see also Table 4.3.).

Beef cattle	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction of total NH_3 emission	29	16	37	18	
Arable	0	0	30	0	
Improved pasture	100	100	60	55.1	
Partially improved pasture	0	0	10	43.8	
Unfenced unimproved pasture	0	0	0	1.6	
Apportioning					
Arable	0	0	11.1	0	**11.1**
Improved pasture	29	16	22.2	9.9	**77.1**
Partially improved pasture	0	0	3.7	7.8	**11.5**
Unfenced unimproved pasture	0	0	0	0.3	**0.3**

Dairy cattle	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	29	16	37	18	
Arable	0	0	30	0	
Improved pasture	100	100	60	57	
Partially improved pasture	0	0	10	43	
Apportioning					
Arable	0	0	11.1	0	**11.1**
Improved pasture	29	16	22.2	10.3	**77.5**
Partially improved pasture	0	0	3.7	7.7	**11.4**

Pigs (sows & fattening)	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	**45**	**13**	**42**	**0**	
Suburban	50	50	0	0	
Arable	50	50	92	0	
Improved pasture	0	0	8	0	
Apportioning					
Suburban	22.5	6.5	0	0	**29.0**
Arable	22.5	6.5	38.64	0	**67.6**
Improved pasture	0	0	3.36	0	**3.4**

Poultry - layers	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	**51**	**10**	**39**	**0**	
Suburban	100	100	0	0	
Arable	0	0	92	0	
Improved pasture	0	0	8	0	
Apportioning					
Suburban	51	10	0	0	**61.0**
Arable	0	0	35.88	0	**35.9**
Improved pasture	0	0	3.12	0	**3.1**

Poultry - broilers	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	**54.5**	**4.5**	**41**	**0**	
Suburban	100	100	0	0	
Arable	0	0	92	0	
Improved pasture	0	0	8	0	
Apportioning					
Suburban	54.5	4.5	0	0	**59.0**
Arable	0	0	37.72	0	**37.7**
Improved pasture	0	0	3.28	0	**3.3**

Poultry - other	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	**53**	**6**	**41**	**0**	
Suburban	100	100	0	0	
Arable	0	0	92	0	
Improved pasture	0	0	8	0	
Apportioning					
Suburban	53	6	0	0	**59.0**
Arable	0	0	37.72	0	**37.7**
Improved pasture	0	0	3.28	0	**3.3**

Sheep	Housing	Storage	Spreading	Grazing	Total weighting
Total	%	%	%	%	%
Fraction	18	16	0	66	
Arable	0	0	0	0	
Improved pasture	100	100	0	57.8	
Partially improved pasture	0	0	0	28.9	
Unfenced unimproved pasture	0	0	0	11.6	
Heather (poorest grazing)	0	0	0	1.9	
Apportioning					
Arable	0	0	0	0	**0**
Improved pasture	18	16	0	38.1	**72.1**
Partially improved pasture	0	0	0	19.1	**19.1**
Unfenced unimproved pasture	0	0	0	7.7	**7.7**
Heather (poorest grazing)	0	0	0	1.3	**1.3**

Emissions from landspreading of livestock manure were distributed over arable land and improved grassland, with a weighted distribution for spreading to each surface type. These rates were derived from the BBSRC (1997a, b) studies, for cattle apportioning 70% of landspreading manures on improved grassland and 30% on arable land, and 8% on grassland and 92% on arable land for pigs. For the spreading of poultry manure, the pig values were adopted.

According to D. Moorhouse (ADAS, pers. comm., 1996) about a third of British pig farmers spread their manure entirely on their own land, while two thirds send some manure further away for spreading. In East Anglia for instance, one of the main pig farming areas in Britain, pigs are raised on straw as a bedding material, which is exchanged with neighbouring cereal farmers for manure. In Humberside, another intensive pig rearing area, there is a large concentration of farms with a very large ratio of pig numbers to farm size. Manure in this area is usually transported further away from its source before spreading. In its basic version described in this chapter, the model was set up to distribute all emissions from manure spreading in the parish they originate from rather than distributing any excess manure outwith the parish after 'saturation' of all suitable land within the parish. This issue and the related spatial uncertainties are discussed further in Chapter 10.

For landspreading emissions from poultry the same assumptions as for pigs had to be made for the basic version of the NH_3 model. Poultry manure from large intensive farms is, however, more 'mobile' than pig manure. According to D. Charles (ADAS, pers. comm., 1996), the majority of manure leaves the farm and is transported for a few miles, e.g. to the next parish. In some areas, such as ‘Sun Valley’ near Hereford, with larger concentrations of intensive poultry farming, the manure will be moved over large distances, in this instance roughly within Herefordshire. Another way of 'disposing' of surplus poultry manure, especially for dried manures, is through incineration in 'poultry power stations' such as at Thetford in East Anglia.

Emissions from housing and manure storage were assumed to occur in close spatial proximity to each other and were therefore distributed jointly. Regarding the spatial location of farmsteads within the parish, the landcover class 'suburban/rural development' was not deemed a suitable medium for general redistribution from all types of farms, as this category is dominated by suburban rather than rural features for large parts of the country. This would have shifted NH_3 emission sources closer to the centres of population than to the more rural locations. For farms dominated by grazing livestock such as cattle and/or sheep, the farmsteads with cattle houses and manure storage facilities are for practical reasons most likely located close to the best grazing land, i.e. improved grassland, rather than any other landcover type. Therefore the housing and storage fractions for cattle and sheep were allocated exclusively to improved grassland, in order to minimise the mis-location of this source of NH_3.

Livestock housing on farms less dependent on grazing land for their livestock, such as large pig and poultry farms, tends to take up more space and therefore is more likely be classified as 'suburban/rural development' in the landcover dataset. This landcover type was taken as the most likely approximation for the spatial location of housing and storage emissions from poultry. Intensive pig farming, although not dependent on grazing land, tends to be linked to and is often located close to arable - especially cereal - farming, which provides cheap bedding material, grain for feedingstuffs and convenient locations for the landspreading of manure accumulated at the farm (D. Moorhouse, ADAS; pers. comm., 1996). It was therefore assumed that the most likely spatial locations for housing and storage emissions of pigs would be approximated by either 'suburban/rural developments' or 'arable land' (Table 5.2.).

For the distribution of grazing emissions from cattle, sheep, goats and farmed deer, a sub-model was established to take varying stocking densities on different quality grazing land into account. For instance, in the rather large parishes in the Highlands of Scotland an even distribution of animals over all land potentially used for grazing would result in a distorted spatial distribution of NH_3 emissions. If such a distribution were used, emissions from nitrogen poor areas such as moorland or heather would be estimated the same as from well fertilised and heavily stocked fields in the more intensively used areas in the glens/valleys. Therefore, grazing emissions were distributed to the different types of pasture according to a percentage factor derived from average annual stocking densities according to agricultural practice (see Table 5.3. below).

Table 5.3. Average annual stocking density values for grazing livestock on different grassland types and % distribution values derived for grazing animals on (grassland) landcover type (source: J. Vipond and B. Lowmon, SAC Edinburgh, pers. comm., 1996)

Livestock class	Landcover type	Avg. annual stocking density	% distribution of grazing animals
Dairy cows	Improved pasture	3 cows ha^{-1}	57.1%
	Partially improved pasture	2.25 cows ha^{-1}	42.9%
	Unfenced unimproved pasture (rough grazing)	-	-
	Total		100%
Other cattle	Improved pasture	1.75 t liveweight ha^{-1}	55.1%
	Partially improved pasture	1.375 t liveweight ha^{-1}	43.3%
	Unfenced unimproved pasture (rough grazing)	0.05 t liveweight ha^{-1}	1.6%
	Total		100%
Sheep, goats & deer	Improved pasture	10 ewes ha^{-1}	57.8%
	Partially improved pasture	5 ewes ha^{-1}	28.9%
	Unfenced unimproved pasture (rough grazing)	2 ewes ha^{-1}	11.6%
	Very poor grazing (heather etc.)	0.33 ewes ha^{-1}	1.9%
	Total		100%

A weighted distribution approach was taken, dependent on the total area of each of the landcover classes available for agricultural use within the parish. Each parish is treated individually and its landcover composition is taken into account. In the example shown in Table 5.4. below, the distribution model for sheep in 2 contrasting parishes, one with relatively poor grazing resources, the other with good pastureland is described in detail. It is assumed that there are 2000 sheep registered in each parish. The number of animals distributed to each landcover class as NH_3 sub-sources is then spread equally over each unit area of the corresponding landcover type on the 1 km modelling grid. The number of hectares (as % cover) of each land cover type in each 1 km grid square is fed into the model from the aggregated landcover maps, together with the total area of each landcover type within the parish to which each gridcell belongs.

Table 5.4. Sheep distribution examples in 2 parishes, P1 and P2.

Parish	Landcover class	% of total area of suitable land in parish (a)	% distribution of sub-sources (Table 5.2. above) (b)*	% of total emission weighted by area of ILC90 classes (c)**	Equivalent to no. of sheep on each landcover type
P1	Improved grass	5%	72%	33%	660
	Part improved	20%	19%	35%	700
	Unimproved	40%	8%	29%	580
	Heather	35%	1%	3%	60
P2	Improved grass	40%	72%	77%	1540
	Part improved	35%	19%	18%	360
	Unimproved	20%	8%	4%	80
	Heather	5%	1%	0.1%	2

*average conditions, not weighted for landcover composition of each parish

**calculated as ([(a) for each landcover category] * [(b) for the same category]) /[Σ all (a)*(b) for the parish]

Assuming that each parish has at least a small percentage of each of the landcover classes needed to distribute all livestock sources, the model should not lose or gain livestock during the redistribution process. There are, however, a number of parishes where one or more of the necessary landcover types is not present in the dataset. In this case, if e.g. there was no improved grassland in the parishes in the example above, the equivalent of 660 and 1540 sheep respectively would be lost through the modelling process.

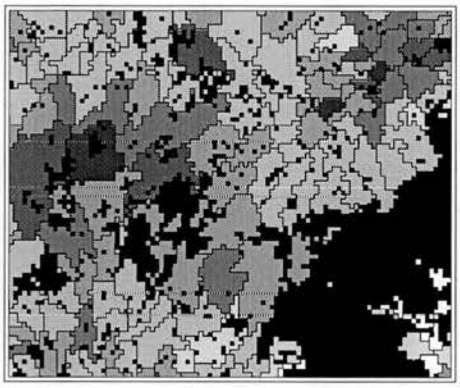
Additionally, a characteristic of classified maps derived from satellite data is that some pixels cannot be classified and are assigned to a separate 'unclassified' category. In the ITE landcover data there are only very few areas which have a large percentage of this category, mainly due to cloudcover (see Figure 4.4.). One of the areas worst affected is the island of Tiree off the west coast of Scotland. A distribution of census items over the parishes affected by this would have resulted in distorted NH_3 concentrations or even entirely empty gridsquares in the case of Tiree.

It was therefore necessary to include rules and conditions into the model to deal with these exceptions. This was achieved by checking for missing landcover types and modifying the percentage distribution factors to each landcover type to account for any missing landcover types. Any potential distribution of census data to non-existent landcover types within each parish was thus reassigned to the next most likely type. For instance, if a parish does not have any arable land, landspreading emissions intended for distribution to arable land are put instead onto improved grassland. The area contribution of the 'unclassified' landclass is added to the last landcover category in the chain for reassignment. This ensures that all NH_3 sub-sources are distributed as sensibly as possible within each parish. The model output was checked thoroughly by re-aggregating all distributed model output and comparing it with the model input at a parish and country level.

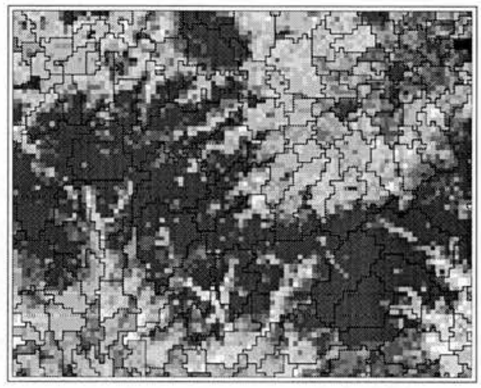
An example of the applied methodology is shown in Figure 5.3. (below) for beef cattle for an area around the Scottish/English border for 1988, comparing the general redistribution by the Edinburgh Data Library with the modified redistribution based on the new landcover dependent weighting for NH_3 emissions. The effect of the modified approach can be summarised as a redistribution of census items as *ammonia sources* as opposed to census items as such.

Using the general methodology developed by Hotson (1988), any gridsquare in, for example, the moorland category would become populated with some livestock types at the same rate as other grid squares which would potentially be used for intensive agricultural activities. This would have the effect of reducing the estimated concentration of livestock in the intensively used areas and increasing the concentration in the extensively used areas. Thus it would not give an accurate reflection of the actual circumstances as it smoothes out the peaks and troughs in the livestock distribution. This aspect is especially important if N deposition and critical loads exceedance models are derived from the results of an NH_3 emissions inventory. The differences between the two methods can be seen clearly in both the 1 km and 5 km maps, with the new approach showing a much greater concentration of emissions in the valleys, as would be expected. This trend is also reflected in the aggregated 5 km grid maps.

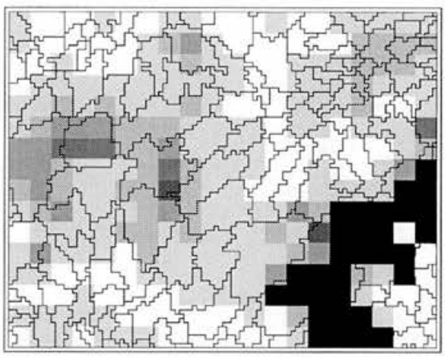
The parish boundaries, which are clearly visible at the 1 km resolution in Figure 5.3a, largely disappear in Figure 5.3b. The discontinuity evident in the general methodology map between England and Scotland (Figures 5.3a,c) is caused by differences in the redistribution rules between England & Wales and Scotland adopted for the general approach at the Edinburgh Data Library. Thus, a sharp contrast appears for some livestock items at the border between the two areas. Using the new methodology, the border is no longer detectable.



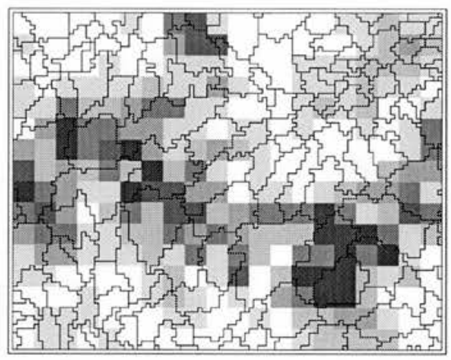
5.3a: Data Library methodology - 1 km grid



5.3b: New redistribution methodology - 1 km grid



5.3c: Data Library methodology - 5 km grid

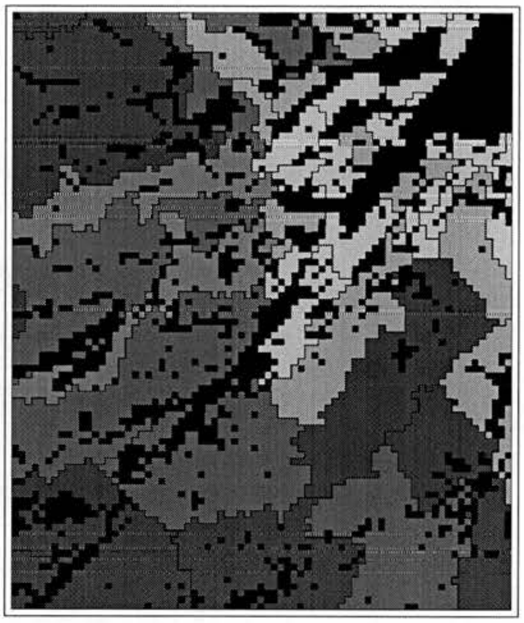


5.3d: New redistribution methodology - 5 km grid

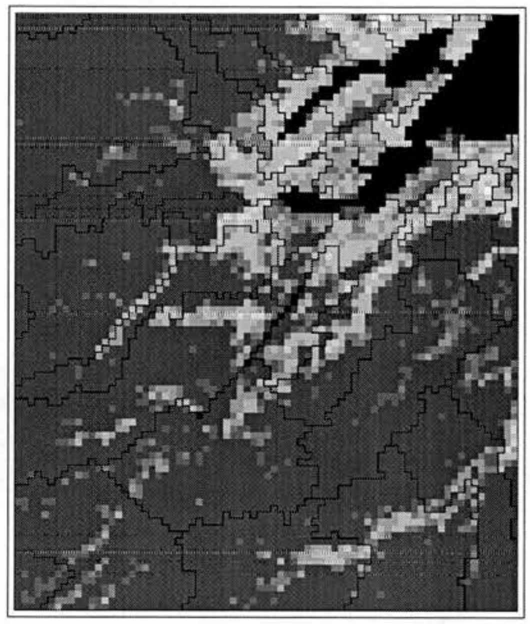
Figure 5.3a-d: Comparison of redistributing beef cattle for an area around the England/Scotland border between the Data Library methodology and the new methodology.

A second example showing the effects of the new methodology is given in Figure 5.4. below for an area of the Grampians, including the Black Isle, in north-eastern Scotland for 1988. In Figures 5.4a and 5.4c the general redistribution of beef cattle by the Data Library method is shown, and this is contrasted with the results of the new NH_3-specific methodology in Figures 5.4b and 5.4d.

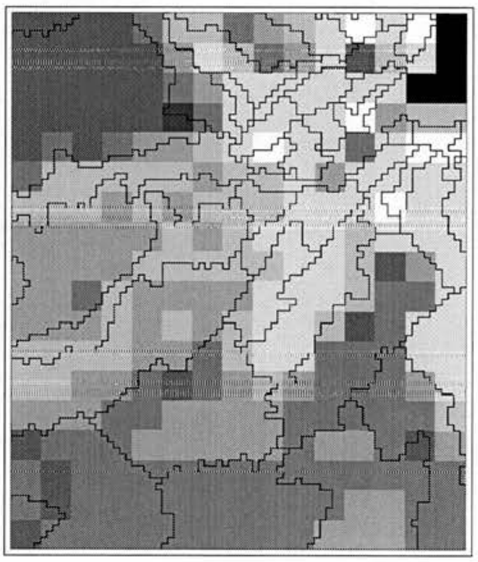
The differences at the 1 km level are dramatic and, again, the parish boundaries clearly visible in Figure 5.4a largely disappear in Figure 5.4b, due to the weighted distribution approach. The more intensive agriculturally used areas in the southern half of the map, a predominantly upland and hill area, such as the Spey Valley (lower right corner) become visible only in the new model.



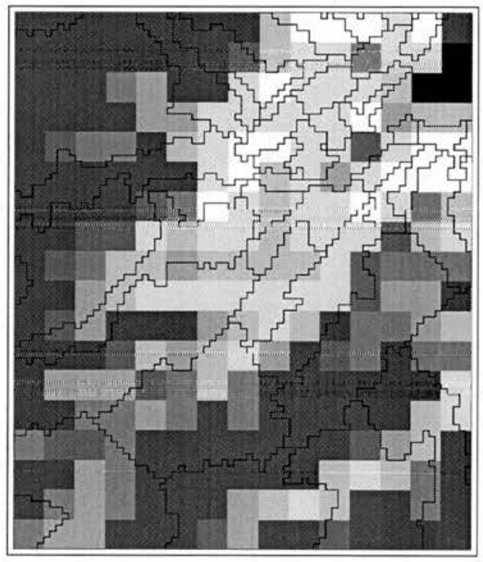
5.4a: Data Library methodology - 1 km grid



5.4b: New redistribution methodology - 1 km grid



5.4c: Data Library methodology - 5 km grid



5.4d: New redistribution methodology - 5 km grid

Figure 5.4a-d: Comparison of redistributing beef cattle for an area of the Grampians (northeast Scotland) between the Data Library methodology and the new methodology.

Figure 5.4. shows some of the largest parishes in the country, with substantial areas of very extensively used land. The application of the new model in this and similar areas results therefore in a considerable spatial relocation of NH_3 sources and consequently of emissions, due to the differentiation between intensive and extensive agricultural landuse.

5.4. THE SPATIAL DISTRIBUTION OF CROPS AND CONSERVED GRASSLAND AS AMMONIA SOURCES

Modelling the spatial distribution of fertiliser N application to crops and conserved grassland as NH_3 sources is a more straightforward process than the livestock distribution. There is no necessity to split the total NH_3 emission estimates for fertilisers into subsources and distribute them over different landcover types. The spatial distribution of fertiliser application as an NH_3 source to different landcover types is unambiguous, as crops receiving fertiliser are allocated to arable land, and fertilised grassland corresponds with the landcover category of 'improved grassland'.

For each parish, the Census data provide hectareages of crops and grassland. The average N fertiliser application rates to the different census items are provided by the annual British Survey of Fertiliser Practice (BSFP). The Agricultural Census and the BSFP data cannot, however, be used directly without converting both to a common classification. In order to match the crop categories of the BSFP and the Agricultural Census, it is necessary to aggregate both the census data and the BSFP tables (see Table 5.5. below).

According to the grassland tables in the BSFP, about one third of all grassland in Great Britain is cut for hay or silage, while the rest is grazed. Since fertiliser emissions from grazed grassland are already included with livestock grazing emissions, double counting had to be avoided. Therefore fertiliser emissions were calculated and redistributed in the model for only one third of the total grass registered in the census. This is again an averaged approximation of the real situation in Britain, with potentially large variations in different parishes or whole regions, depending on the local practice.

In order to make the model more efficient, the crops and crop groups data were not distributed separately, but aggregated further by calculating the total amount of fertiliser applied within the parish to all crops. This poses no problems as all the crops and crop groups are distributed over only one landcover type. Only fertiliser applications to grassland were treated separately, as they were distributed over a different landcover type.

Table 5.5. Fertiliser application rates to aggregated crops, crop groups and grassland (after Burnhill *et al.*, 1997 and census data for 1996).

Census items (1996)	England & Wales item no.	kg N ha^{-1}	Scotland item no.	kg N ha^{-1}
Wheat	11	185	14	190
Winter barley	12	138	16	162
Spring barley	13	95	18	93
Oats	14	125	17,20	111[a]
Rye	16	126	-	-
Potatoes	19	174	25,26	140
Sugarbeet	20	107	-	-
Oilseed rape	29,36	190	19,23	175
Linseed	30	53	21	51
Forage maize	17	52	-	-
Turnips for stockfeeding	24	57	29	73
Kale & cow cabbage	26	93	30	100
Other roots and green crops	25,28	80	31,32,34	112
Peas	27,195,196	2	28	2
Beans	23,190,192	7	27	7
Vegetables - brassicae	170,172-175	201	-	-
Other vegetables	178,181,182,185,186,197-200	96	35[b]	114[b]
Small fruit	226-(207+208)	78	36,37[c]	64[c]
Top fruit	227,228	50	-	-
Other tillage[d]	15,21,31,37,205,236,244	63	15,22,38	32
Grass	5,6	149	42-45	149

[a] value for GB used here, due to a very small sample size in Scotland, [b] all vegetables in Scotland, [c] all fruit in Scotland, [d] all other crops from the census data which did not have any straight matches in the BSFP.

In the same manner as for livestock, conditional rules were built into the redistribution model for fertiliser emissions to ensure that all NH_3 sources were distributed without losses due to the composition in the landcover data. This was, however, less of an issue than with the livestock model, with the exception of Tiree, due to the cloudcover problem (see Section 5.3.)

5.5. METHODOLOGY FOR THE CALCULATION OF AMMONIA EMISSIONS FROM LIVESTOCK HUSBANDRY AND MINERAL FERTILISERS

After redistributing all census items as NH_3 sources over the landcover data, the next step is the calculation of the spatial emissions inventory. This is accomplished in a relatively straightforward manner, through the calculation of emissions as a product of average source strength estimates with the grid matrices containing the spatially distributed source data. The latter contain the number of sources, i.e. the number of animal equivalents in each livestock category and the total amount of mineral N fertiliser applied to crops, for each grid square in the model domain. The model calculates intermediate inventories for all source categories, which are subtotalled to NH_3 emissions from all livestock sources and NH_3 emissions from all fertiliser applications. These are further summarised to total emissions from all agricultural sources.

All three end products of the model, the livestock, fertiliser and total emissions maps are aggregated to the 5 km grid level for mapping/publication. This reduces the spatial uncertainty resulting from the assumptions and rules in the model as well as errors and uncertainties in the input datasets (Figure 5.5.). While the temporarily (during run-time of the model only) separate inventories for sheep, cattle, pigs etc. as well as all results at the 1 km level would potentially be disclosive for the 1996 census, the model output aggregated to the 5 km level and the 3 source categories described above satisfies all the rules regarding disclosivity as agreed with MAFF and SOAEFD.

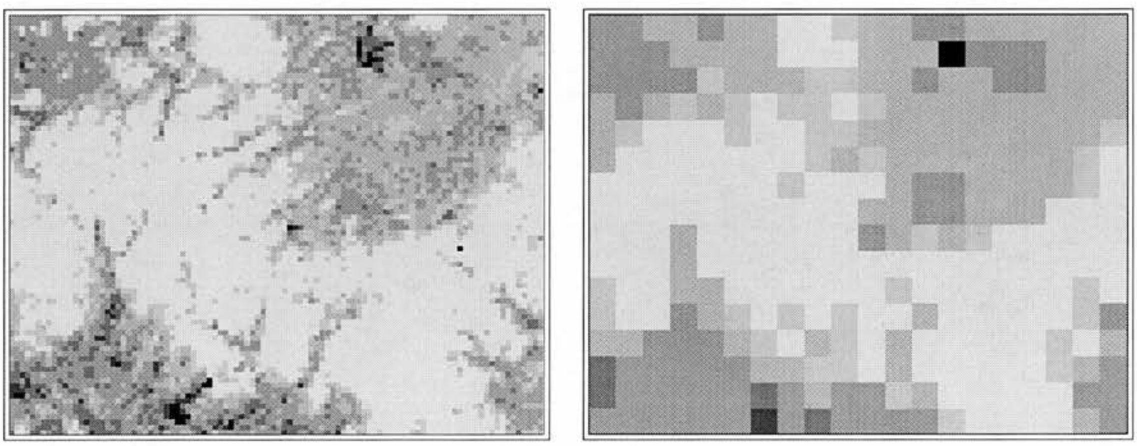


Figure 5.5. Total NH_3 emissions from all agricultural sources in an area in the Scottish Borders at an a) 1 km and b) 5 km resolution for 1988. Dark areas indicate high NH_3 emission level.

The above method was also applied to the already spatially distributed data for Northern Ireland. The only difference here was that the resolution of the spatially distributed NH_3 sources was already at the 5 km level. Therefore no further aggregation of the results was required after the model runs to preserve confidentiality and non-disclosivity.

With the methodology in place, scenarios using different emission source strength estimates can be calculated. This includes, for instance, the estimation of reductions from abatement measures applied to certain categories of livestock, as discussed in Section 7.4. Furthermore, alternative scenarios can be created by applying NH_3 source strength data from inventories other than the UK 'official estimate' (DoE, 1995; RGAR, 1997), and the present model could easily be refined to represent livestock subcategories in more detail. These issues regarding model sensitivity testing and uncertainties are discussed further in Chapter 9.

5.6. METHODOLOGY FOR THE CALCULATION OF AMMONIA EMISSIONS FROM NON-AGRICULTURAL AND OTHER MISCELLANEOUS SOURCES

Non-agricultural NH_3 emissions originate from a multitude of relatively small sources, as shown in Section 3.4. Some of these are difficult to pinpoint spatially without surveys (see also Table 5.6. below).

Emissions linked to the human population are easier to locate, as mapped population numbers are available from the UK Population Census. This survey is carried out every 10 years, with trends calculated annually for the period between two censuses (e.g. Great Britain Office of Population Censuses and Surveys, 1998). For this study, the data for 1981 were made available at a 10 km by 10 km grid resolution (J. Goodwin, AEA Technology, Culham, pers. comm., 1995). The 1981 dataset was brought up to date regarding the total number of people living in the UK by scaling it according to the 58.6 million people estimated for 1996 (see Figure 5.6.). This does not take account of any population movement within the UK over the 15 year period since the data were collected. However, compared with the scale of all the other uncertainties involved in the spatial distribution of non-agricultural emissions, this is probably one of the smallest factors contributing to the overall spatial uncertainty.

The spatial location of other NH_3 sources such as seabirds, wild animals, or landspreading of sewage sludge had to be approximated by linking them to the landcover data. The majority of seabird emissions occur close to the sea. Wild animals are mostly found in natural and semi-natural habitats, but also on agricultural land and in suburban and rural areas. Areas where straw and stubble burning are likely to occur are easier to locate, as this practice is mainly limited to arable fields. Sewage sludge spreading is most likely to take place on agricultural land and also in some woodland areas. It is, however, difficult to distinguish between areas/regions where this practice is common and others where it is not.

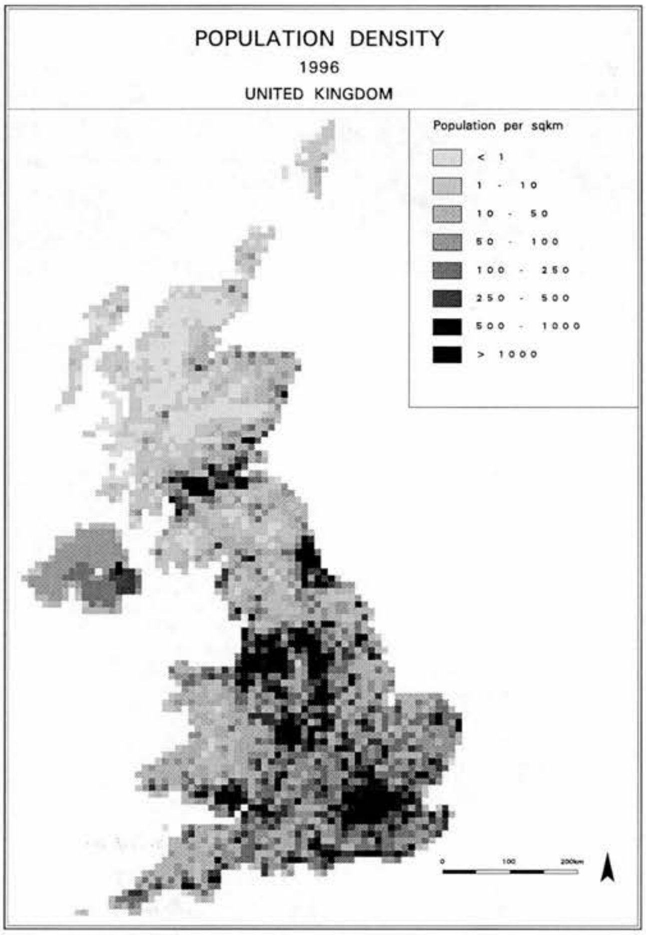


Figure 5.6. The spatial distribution of the UK population according to the Census of 1981, adjusted to population estimate for 1996.

Industrial sources can theoretically be pinpointed individually on the map, provided source strength information is available for every location where NH_3 is emitted. Some information which would be extremely valuable for this purpose has recently become available for England and Wales from the Environment Agency's Chemical Release Inventory (CRI). This database provides detailed emissions on an installation basis for all processes requiring authorisation. The reported numbers are a combination of measurements and estimates. The only industrial process for which coordinates and individual NH_3 emission estimates were available for this study is the processing of sugarbeet.

Table 5.6. Spatial distribution of non-agricultural NH_3 emissions for the UK (after Sutton *et al.*, 1998a)

NH_3 source type	spatial allocation
direct human emissions (breath, sweat, smoking, nappies)	scaled by population
horses	scaled by population
pets (cats & dogs)	scaled by population
seabirds	coastal areas (cliffs, beaches, coastal bare land, salt marshes etc.)
wild animals (deer, rabbits, etc.)	natural and semi-natural land, agricultural land, suburban areas
biomass burning	arable land
sewage works	scaled by population
sewage sludge spreading	agricultural land and coniferous woodland
transport	scaled by population
landfill sites	scaled by population
industrial sources (except sugarbeet processing)	scaled by population
sugarbeet processing	individual locations & emissions available
coal combustion (domestic & industrial)	scaled by population
waste incineration	scaled by population
household products	scaled by population

The best available approximations to model the distribution of emissions from these miscellaneous sources are summarised in Table 5.6. These were applied here, which results in overestimates of NH_3 emissions from industrial sources and horses in densely populated areas, especially around London. Further work is required to reduce the bias towards populated areas in this estimate and hence the spatial uncertainty of this estimate.

5.7. SUMMARY AND CONCLUSIONS

The spatial pattern of NH_3 emissions is a function of the distribution of NH_3 source activities, mainly of agricultural livestock, crops and grassland as well as other miscellaneous sources. Previous spatially distributed emission inventories of agricultural NH_3 were developed on the basis of existing general spatial distributions of agricultural census data. This led to an overestimation of NH_3 emissions in extensively farmed upland areas and an underestimation of emissions in intensively used lowland areas.

This chapter introduces a new approach which specifically distributes NH_3 sources in a way that reflects real circumstances more closely, rather than simply the general sector activity. Component emission sources such as livestock grazing, housing and manure storage as well as landspreading and fertiliser application are spatially redistributed from agricultural census data on a parish basis to a regular grid. The model performs a weighted spatial distribution of livestock and fertiliser emissions through the integration of landcover data, by locating the component emission sources on suitable landcover types. This includes a submodel for livestock grazing emissions, which takes account of varying stocking rates for different quality grazing land, from well-fertilised lowland pastures to rough grazing in the uplands and hills. In a second step, the agricultural NH_3 emission inventory is calculated from the distributed sources, by applying emission source strength estimates.

The method is expected to be much more realistic, compared with previously applied 'general distribution' approaches. Previous methods took no account of a) the varying density of sources on different landcover types and b) the varying NH_3 emissions probability for different source activities (housing, manure storage, manreu spreading, livestock grazing).

Emissions from other miscellaneous sources such as humans, wild animals and birds, industry, transport etc. were distributed via a simpler approach. The estimated total emissions from each source category were either scaled by spatially distributed population census data, or spread proportionally over suitable landcover types.

Chapter 6
Application of the new model to describe ammonia emissions for the UK

6.1. INTRODUCTION

In the following sections, the new spatially distributed UK NH_3 emissions inventory is described in detail. The results are summarised separately for the different source groups: agricultural livestock, fertiliser use on crops and conserved grassland, non-agricultural emissions, as well as total NH_3 emissions. Model output for 1988 is compared between the old methodology (Hotson, 1988) and the new methodology developed here. The absolute and relative importance of the different livestock sub-sources, such as cattle, sheep etc. is also investigated, regarding their contributions to the total emissions (Section 6.6.). Furthermore, the results of this study are compared with those of other, earlier studies (e.g. Kruse, 1986; Eager, 1992; Sutton *et al.*, 1995; see Section 6.5.).

In Sections 6.2.-6.6., the results of the detailed analysis of the spatial distribution of NH_3 emissions are described for Great Britain for 1988 only, to provide a consistent overview between the different source categories and their contributions to the overall NH_3 emissions. The spatially distributed results for agricultural emissions for 1996 are discussed in Section 6.7. for Northern Ireland and in Chapter 7, which considers temporal changes. In this chapter, detailed spatial analyses are shown for 1988 rather than the more recent 1996 inventory. This is necessary as presenting results for livestock classes separately would potentially violate the rules set out for the use of the 1996 census data in the disclosivity agreement with MAFF and SOAEFD for some gridsquares (see also Section 4.2.1.).

Throughout Chapter 6, agricultural emission estimates were calculated following the source strength estimates of DoE (1995). The total emissions for 1988 are summarised in Table 6.1. and compared with the estimates for 1996. Results of a model sensitivity analysis using other authors' source strength estimates (BBSRC, 1997b; TFEI, 1996) are discussed in Chapter 9. In the absence of specific estimates for 1988, the 1996 estimates of Sutton *et al.* (1998a) were applied for non-agricultural sources for both years (Table 6.1.).

The new methodology of spatially redistributing NH_3 emissions developed in the previous chapters does not change the total sums for Great Britain, compared with the tabulated version below (Table 6.1.). The emission source strength estimates per unit livestock or per kg N fertiliser ha^{-1} are still applied as averages for the whole country in the new methodology developed in this study (Chapters 4-5). Several approaches which are likely to change emission source strength estimates and thus total emissions for parts of the UK are outlined in Chapters 9-10. An approach introducing spatially variable emission source strength data into future versions of the model would not only change the magnitude of emissions in local areas, but is likely to have an impact on the total magnitude of UK emissions, due to non-linearity issues.

Table 6.1: Estimated NH_3 emissions from agricultural and other miscellaneous sources in the UK 1988 and 1996 [Notes: [a]UK Agricultural Census 1988: GSS. (1993), [b]UK Agricultural Census 1996: sum of model input data, [c] estimated for 1996; NH_3 emission source strength data: DoE (1995) for agricultural sources and Sutton *et al.* (1998a) for other miscellaneous sources].

Category	Animals UK 1988[a]	Animals UK 1996[b]	kt NH_3-N UK 1988	kt NH_3-N UK 1996	Contribution % 1988	Contribution % 1996
Cattle	11,902,000	11,904,000	133.7	133.7	45.3%	45.5%
Sheep, goats	41,028,000	41,623,000	15.6	15.9	5.3%	5.4%
Pigs	7,983,000	7,506,000	25.4	23.9	8.6%	8.1%
Poultry	132,866,000	146,496,500	25.2	27.8	8.5%	9.5%
Deer	n/a	33,700	0.03[c]	0.03	0.01%	0.01%
Total livestock	-	-	199.9	201.3	67.7%	68.5%
Fertiliser	-	-	32.5[a]	29.7[b]	11.0%	10.1%
Total agriculture	-	-	232.4	231.0	78.7%	78.6%
Misc. sources	-	-	62.9[c]	62.9	21.3%	21.4%
Total	-	-	295.3	293.9	100%	100%

Since the atmospheric lifetime of gaseous NH_3 is rather short, large gradients of NH_3 deposition occur downwind of sources on spatial scales of < 10 km as well as at the field scale (see also Chapter 8). The improved accuracy of the spatial distribution of NH_3 emissions achieved by the new methodology is especially important for defining source and sink areas regarding NH_3 deposition. Deposition and impacts of NH_3 in the UK were modelled using the results of this study (e.g. Singles, 1996; Singles *et al.*, 1998; Sutton *et al.*, 1998c) and are discussed in Chapter 11.

6.2. SPATIAL DISTRIBUTION OF EMISSIONS FROM AGRICULTURAL LIVESTOCK

6.2.1. Comparison of the two model approaches for livestock emissions in 1988

Ammonia emissions from agricultural livestock contributed 86% and 87% of the total agricultural emissions in the UK in 1988 and 1996, respectively, using the NH_3 source strength data of DoE (1995) for livestock and fertiliser sources (Table 6.1.). This amounts to about 68% of the total UK NH_3 emissions including other miscellaneous sources (after Sutton *et al.*, 1998a), for both years. The spatial distribution of these livestock emissions was modelled for both years, using the new methodology described in Chapters 4 and 5.

Additionally, the simpler methodology used by earlier studies (e.g. Kruse, 1986; Eager, 1992; Sutton *et al.*, 1995; Dragosits *et al.*, 1996b) for developing spatial NH_3 emission inventories was applied to already spatially distributed agricultural census data for 1988 (see Section 5.5.). These data were supplied by the Edinburgh University Data Library, who applied the method by Hotson (1988) to the 1988 parish census data (Sections 5.1., 5.2.). In this simplified methodology the emission source strength data were applied to the redistributed census data without taking account of the spatial distribution of the sources themselves.

The two NH_3 emission inventories derived for 1988 were thus compared, using the same emission source strength data and the same original parish census data (see Figures 6.1a and b; Sections 5.1. and 5.2.):

- 'Old model' - inventory with agricultural census data redistributed following the approach by Hotson (1988);
- 'New model' - inventory with agricultural census data redistributed as NH_3 sources following the new approach developed in this study.

The inventories are compared in Figure 6.1. for total livestock emissions. In addition, the results of the new model are shown in greater detail in Section 6.2.2., distinguishing the major livestock source categories: cattle, sheep, pigs and poultry.

In the new model, NH_3 emissions from livestock have been concentrated in areas which are more suitable for intensive agricultural activities within each parish, rather than spread evenly over all land used for agricultural purposes (see also Sections

5.1.-5.3.). Areas with little agricultural activity such as moorland, heathland and other semi-natural vegetation types have had NH_3 source density and therefore emissions reduced, compared with the old model, and sources were moved to more intensively used lowland areas. The conditions and rules specified in the new model were developed to mirror the reality of agricultural practice, regarding NH_3 source distribution, rather than a more general distribution of livestock (see also Figures 5.3. and 5.4.). In Figure 6.1b, this is especially noticeable in the Scottish/English Borders area, where emission sources and therefore emissions have been moved off the extensively grazed rough hill pastures. Thus emissions have been concentrated at the foothills and in the valleys, which can be identified much more clearly. Further areas with prominent decreases in emission are the Pennines, the Cumbrian hills and some areas of the Welsh hills.

It should be noted that both the new and the old model redistribute livestock census data only within the boundaries of each parish. Thus not only the distribution of landcover types within each parish, but also the size and shape of the individual parishes play an important role in determining how far livestock can potentially be moved within the model. For smaller parishes, the new model is estimated to show less differences in emission source distribution than for larger parishes. The better performance of the model in upland and hill areas is therefore partially linked to the generally larger parish sizes and the specific structure of parishes found in these areas. For instance, parishes in the Borders and Highlands of Scotland (compare Figures 5.3b; 5.4b) tend to contain a part of the better lowland pastures in the glens and at the foot of hill areas, and equally share the extensive rough grazing land of the moorland and hill areas (e.g. Rackham, 1986).

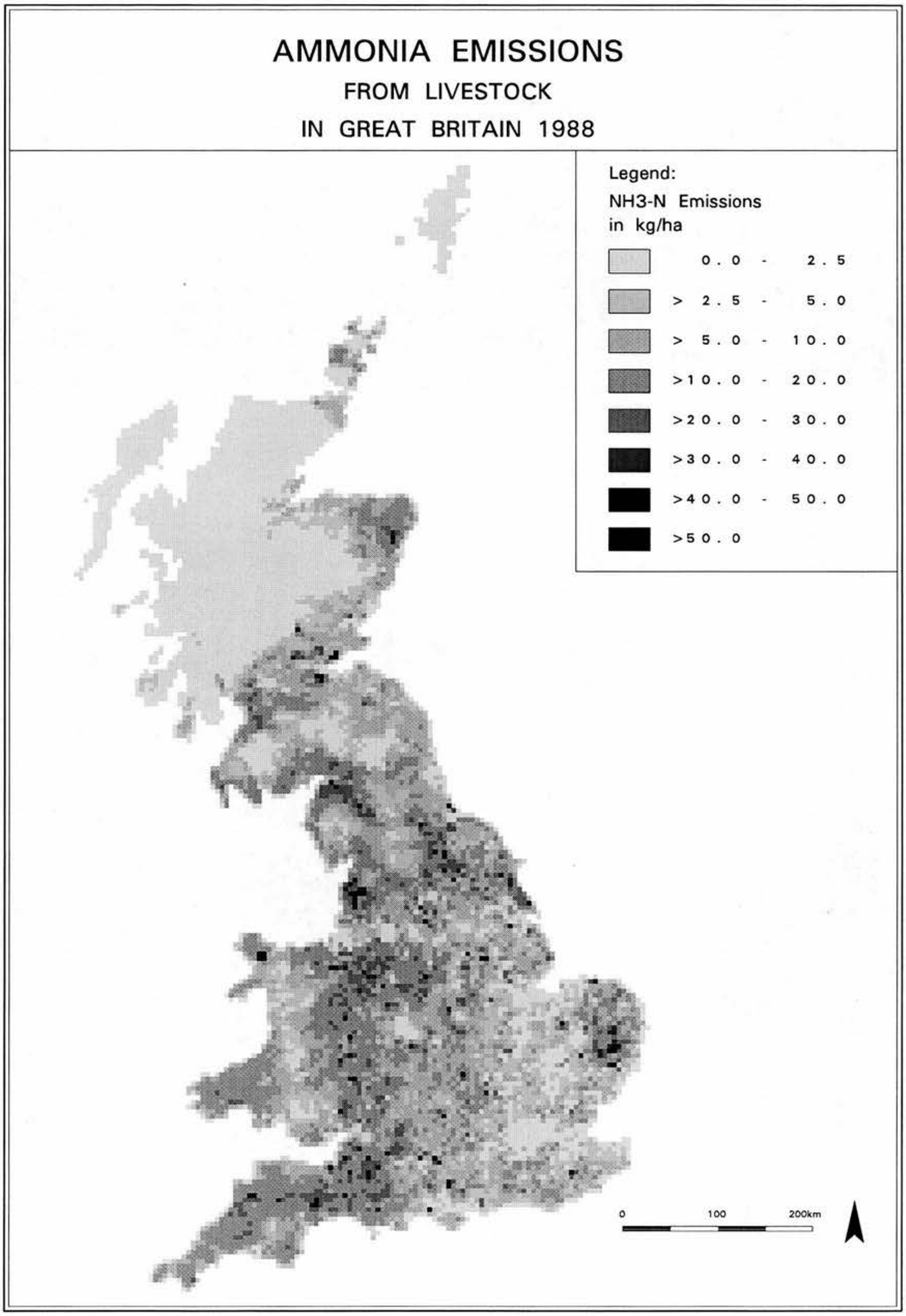


Figure 6.1.a: Total NH_3 emissions from livestock sources in Great Britain 1988 at a 5 km resolution using the old census redistribution model.

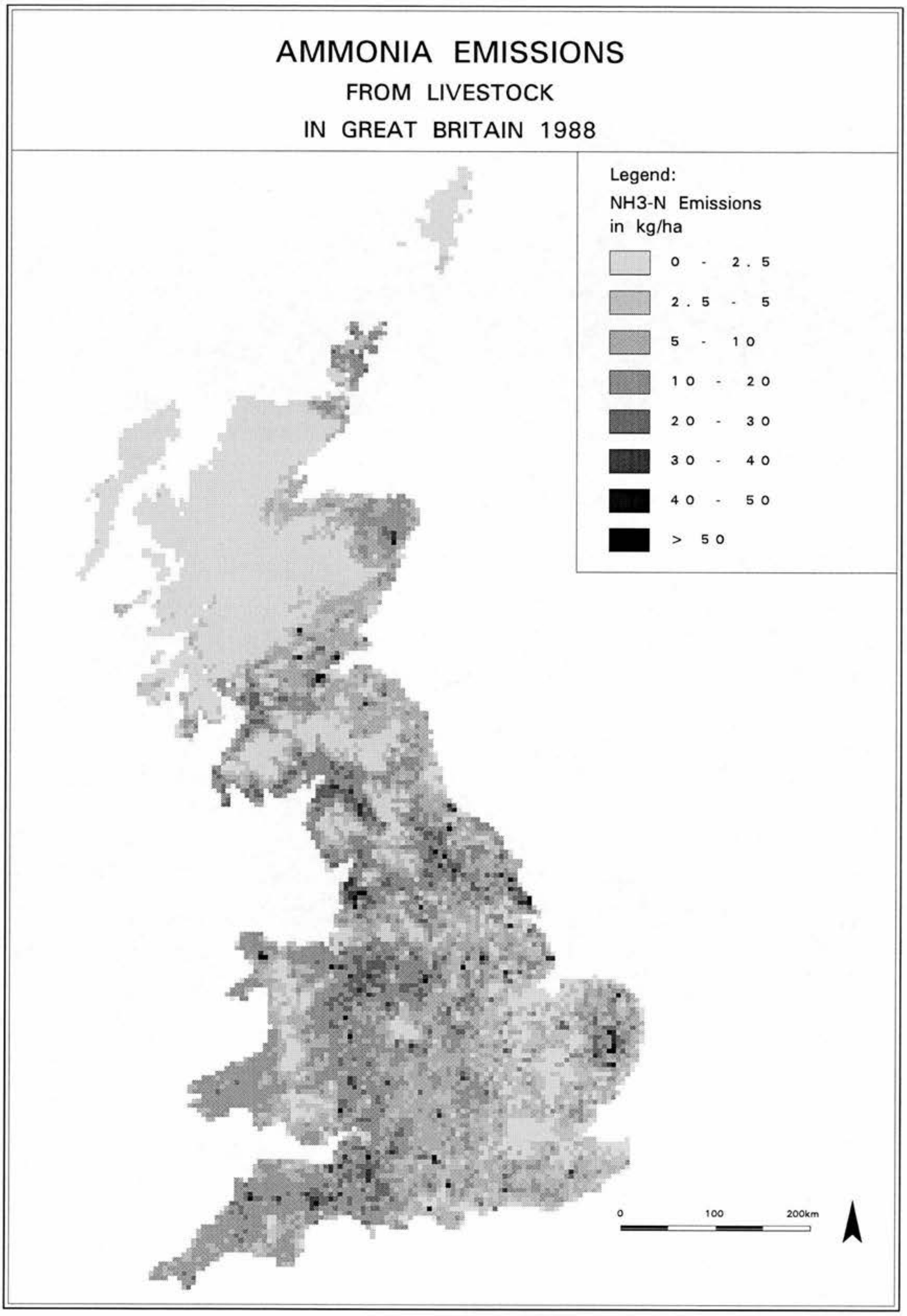


Figure 6.1.b: Total NH_3 emissions from livestock sources in Great Britain 1988 at a 5 km resolution using the new census redistribution model.

6.2.2. Spatial distribution of emissions from livestock source categories in 1988

A simple geostatistical analysis of the NH_3 emission maps from agricultural livestock for Great Britain was carried out to determine the spatial distribution of different levels of NH_3 emissions (Table 6.2.). The mean value of livestock emissions over all grid squares for Great Britain was 8 kg N ha^{-1} $year^{-1}$. This is, for instance, equivalent to approximately 1500 cattle and 500 pigs in a 5 km gridcell.

Table 6.2. shows that nearly one third of all grid squares (31%) are in the lowest emission category with less than 2.5 kg N ha^{-1} from livestock sources on average. These values are typical for extensive upland and hill areas (e.g. most of the Scottish Highlands and Islands) as well as urban areas (e.g. Greater London).

Table 6.2. Analysis of NH_3 emissions from agricultural livestock for 1988: % of 5 km grid squares per category.

Ammonia emission category (kg ha^{-1} NH_3-N)	proportion of GB grid squares
>0-2.5	31.2%
>2.5-5	14.8%
>5-10	24.2%
>10-20	24.3%
>20-30	4.4%
>30-40	0.7%
>40-50	0.2%
> 50	0.2%

Typical values for about half of the 5 km grid squares of Great Britain are in the 5-20 kg ha^{-1} range. In total, 95% and 98% of all grid squares fall into the categories below 20 kg ha^{-1} and 30 kg ha^{-1}, respectively. The highest values in the model output with emissions of above 30 kg N ha^{-1} occur in only 1% of all grid squares, with only 0.2% above 50 kg ha^{-1} (maximum value 115 kg ha^{-1}).

The grid squares affected by the largest emission concentrations in Table 6.2. all contain a large contribution from intensive farming of pigs and/or poultry (compare Figures 6.1b, 6.2c and 6.2d). This highlights the local importance of intensive pig and poultry farming. In total, pig and poultry farming contributed only 12.7% and 12.6% respectively to the total livestock emissions in the UK in 1988 (Table 6.1.). Although the total magnitude of NH_3 emissions from pigs and poultry in the UK is relatively small, compared with cattle at 67% (see Table 6.1.), these sources are associated with high emissions in individual grid squares (see also Section 6.6.).

In a spatial context, intensive poultry farming is in many cases associated with localised high emission sources in close proximity to population centres (compare

Figures 5.6., 6.2d), whereas large pig farms are more regionally based. The main centres of intensive pig farming are located around Yorkshire and Humberside as well as in East Anglia. Coppock (1976a) associates the clustering of pig producing enterprises in these areas with the presence of intensive vegetable and crop production. Pigs are mostly fed on concentrates, surplus and low quality crops or crop residues. Intensive pig production is also often associated with smaller holding sizes (by area), as they are mostly kept in artificial environments and not dependent on the availability of grazing land.

Grass-based livestock farming, i.e. cattle and sheep farming, on the other hand, are less intensive per unit area of the holding. This is reflected in the associated NH_3 emissions and their spatial distribution: both cattle and sheep emission sources are more evenly distributed than the more localised intensive poultry or pig sources. Overall, cattle and sheep emissions together account for about three quarters of the total livestock emissions for 1988, with cattle contributing 67%, and sheep 8% (see Table 6.1.).

The areas with the highest emissions from cattle are the lower lying and more fertile areas (Figure 6.2a) of Britain, with intensive dairy farming concentrated in three major areas: Cheshire and the surrounding counties, north-west England, and south-west Wales, Dorset, Somerset and the West Country.

Sheep emissions are highest in Wales, northern England and southern Scotland. The high concentrations of sheep in Wales and the related high sheep emission estimates (Figure 6.2b) are partially due to the better quality grazing compared with other upland areas, and because more use is made of 'in-bye land' which allows higher carrying capacities. Additionally, the principal sheep breed in Wales is the Welsh Mountain Sheep, one of the smallest British sheep breeds, which can be stocked more densely than the larger sheep more common elsewhere (see also Section 2.4.1.). This fact is likely to cause an overestimate of sheep emissions for Wales compared with other areas, as the smaller sheep breeds are estimated to have lower N excretion rates and thus also smaller NH_3 emissions. This also supports the argument for using spatially varying emission source strength estimates in emission inventories, where sufficient data are available.

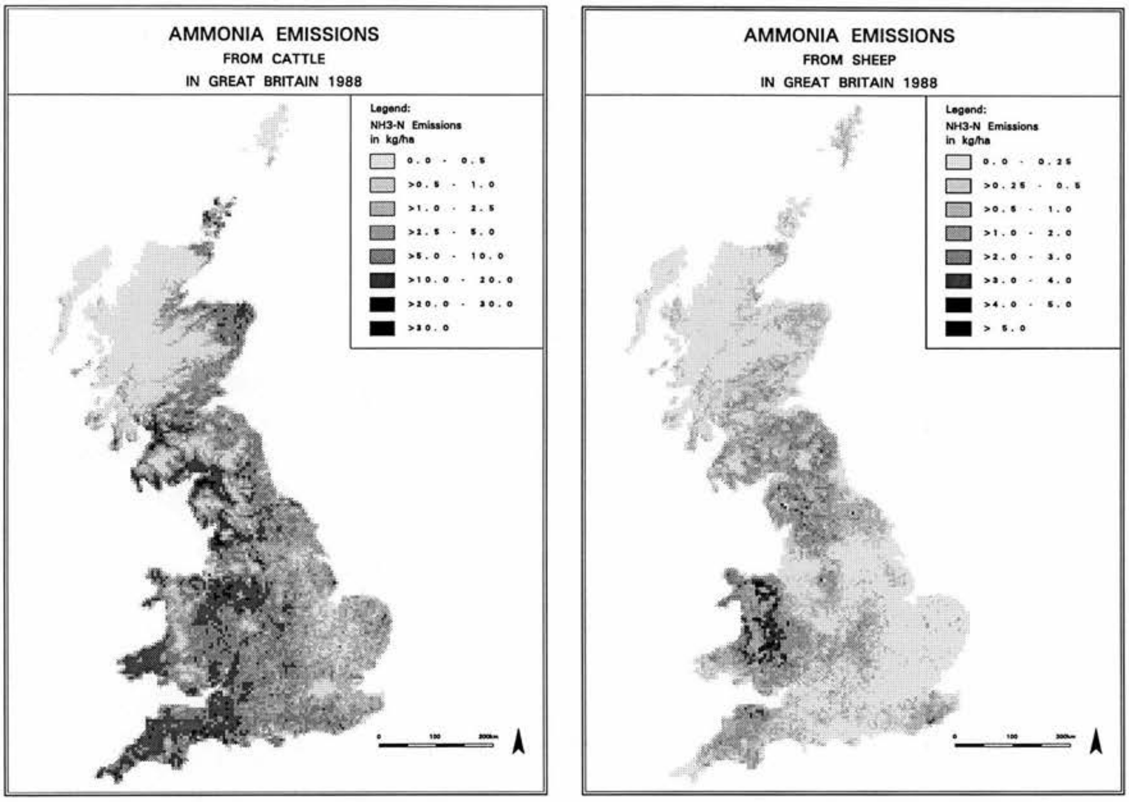


Figure 6.2. a-b: Ammonia emissions from livestock sources in Great Britain 1988 at a 5 km resolution a) emissions from cattle; b) emissions from sheep (new census redistribution model).

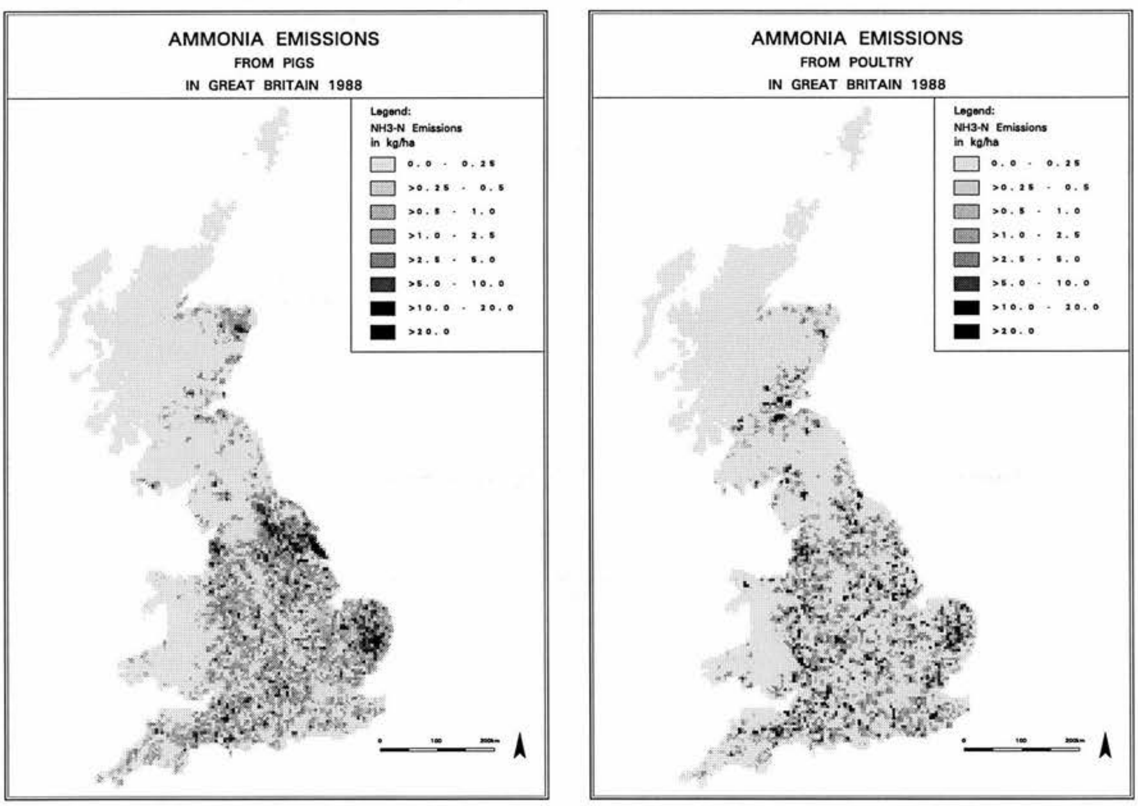


Figure 6.2. c-d: Ammonia emissions from livestock sources in Great Britain 1988 at a 5 km resolution c) emissions from pigs; d) emissions from poultry (new census redistribution model).

In areas with low total livestock emissions, such as the Highlands and Islands of Scotland or the upland areas of England and Wales, livestock emissions, especially from sheep and cattle, still provide the largest contribution to emissions. The importance of the different livestock categories as emission sources in each grid square and their differing spatial patterns are discussed further in Section 6.6.

6.3. SPATIAL DISTRIBUTION OF EMISSIONS FROM MINERAL FERTILISER APPLICATION AND CROPS

Ammonia emissions from fertiliser application contributed about 32.5 kt NH_3-N to the UK emissions in 1988, which is equivalent to 14% of the agricultural NH_3 emissions and 11% of the total NH_3 emissions, respectively (Table 6.1.). This constitutes only a relatively small fraction of the total emissions, but nevertheless plays a significant role in some areas of the country (see Figure 6.3.). Fertiliser emissions are especially important in the eastern half of England, but also, to a lesser degree, in central and eastern parts of Scotland. These areas represent the main crop production areas and are intensively managed with large N fertiliser application rates.

The grassland areas, which are a more prominent feature in the western half of England, in Wales and some parts of eastern Scotland, also show some regions with relatively high fertiliser emissions, but generally appear less prominent (Figure 6.3.) than the areas with predominantly arable crops. This is due to fertiliser emissions from grazed grassland being included with livestock grazing emissions; only emissions from fertilisers applied to conserved grassland were mapped in Figure 6.3., to avoid double counting. Higher fertiliser emissions from conserved grassland are generally spatially linked with higher proportions of improved grassland in areas of dairying and intensive beef and sheep rearing.

A large proportion of grid squares (see Table 6.3., Figure 6.3.) shows relatively small fertiliser emissions per hectare. Nearly 50% of all squares in Great Britain are estimated to have average emissions of less than 1 kg N ha^{-1} from fertiliser, which is equivalent to average fertiliser application rates of under 35 kg N ha^{-1} within the grid square. This is not a very likely scenario, as most crops and grasslands receive between 100 and 200 kg N fertiliser ha^{-1} on average (BSFP, 1997). The reason for this is that any single grid square is not completely covered by arable crops and conserved grassland. A more realistic scenario would be, for instance, a quarter of the grid square (625 ha) receiving on average 136 kg ha^{-1} fertiliser, with the rest (1875 ha) being occupied by grazed grassland, built-up area, woodland, etc.

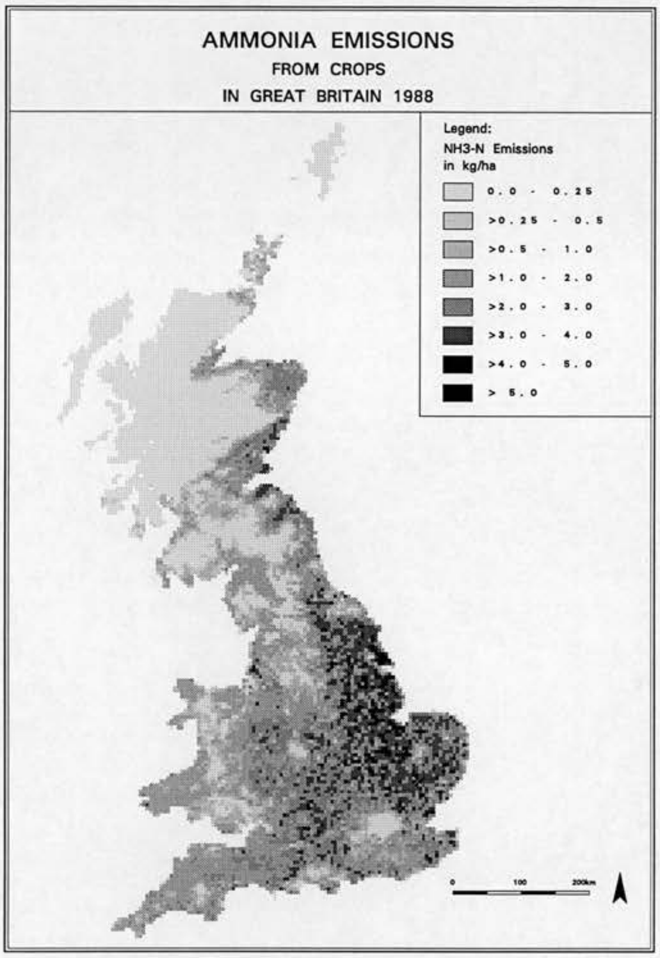


Figure 6.3. Ammonia emissions from fertiliser application to crops and conserved grassland for Great Britain in 1988; using the new model.

Only 2.5% of all grid squares are estimated to have emission rates of over 4 kg ha^{-1}, with 4 kg being equivalent to 194 kg ha^{-1} over 70% of the area. These estimates are reasonable for the mainly arable areas in the eastern half of England (see Figure 6.3.), thus showing that the redistribution model does not appear to cause unrealistically high emission totals for any squares. The average emission estimate from fertilisers in Great Britain, however, is relatively low at 1.3 kg ha^{-1} for 1988, which reflects the relative importance of pastureland of varying quality and other landuse types such as agriculturally unproductive land in large parts of the UK.

Table 6.3. Analysis of NH_3 emissions from mineral N fertiliser application and crops for Great Britain in 1988: % of 5 km grid squares per category.

Ammonia emission category (kg ha^{-1} NH_3-N)	proportion of GB grid squares
>0-0.25	29.6%
>0.25-0.5	6.2%
>0.5-1	12.4%
>1-2	25.0%
>2-3	15.7%
>3-4	8.6%
>4-5	2.1%
>5	0.4%

As outlined in Section 5.4., there is minimal difference between the spatial patterns of NH_3 emissions from fertiliser between the simple model and the new model of source redistribution. This is due to the similarity of the landcover data and redistribution rules in both models for these more 'stationary' sources, compared with livestock emissions, which apply to many different landcover types.

6.4. SPATIAL DISTRIBUTION OF AMMONIA EMISSIONS FROM NON-AGRICULTURAL SOURCES

Ammonia emissions from sources other than agriculture, i.e. humans, horses, pets, industry, transport, combustion etc. (Table 3.8.), contributed a substantial component to the total UK emissions in 1988 and 1996. The estimated total of 62.9 kt NH_3-N (Sutton *et al.*, 1998a) constitutes approximately 21% of the NH_3 emitted in the UK, using the DoE (1995) source strength estimates for agricultural sources (Table 6.1.).

Figure 6.4. shows the estimated spatial distribution of these miscellaneous sources. The pattern resulting from the allocation rules of the model is much cruder and much more uncertain than that of the agricultural sources. Nearly two thirds of the total non-agricultural emission sources (40.7 kt N $year^{-1}$) had to be scaled and spatially distributed by human population numbers as the best simple approximation, and others distributed evenly over landcover types where these sources were most likely to be found. The uncertainties resulting from this simplified redistribution approach are further discussed in Chapter 9, together with suggestions for improvements.

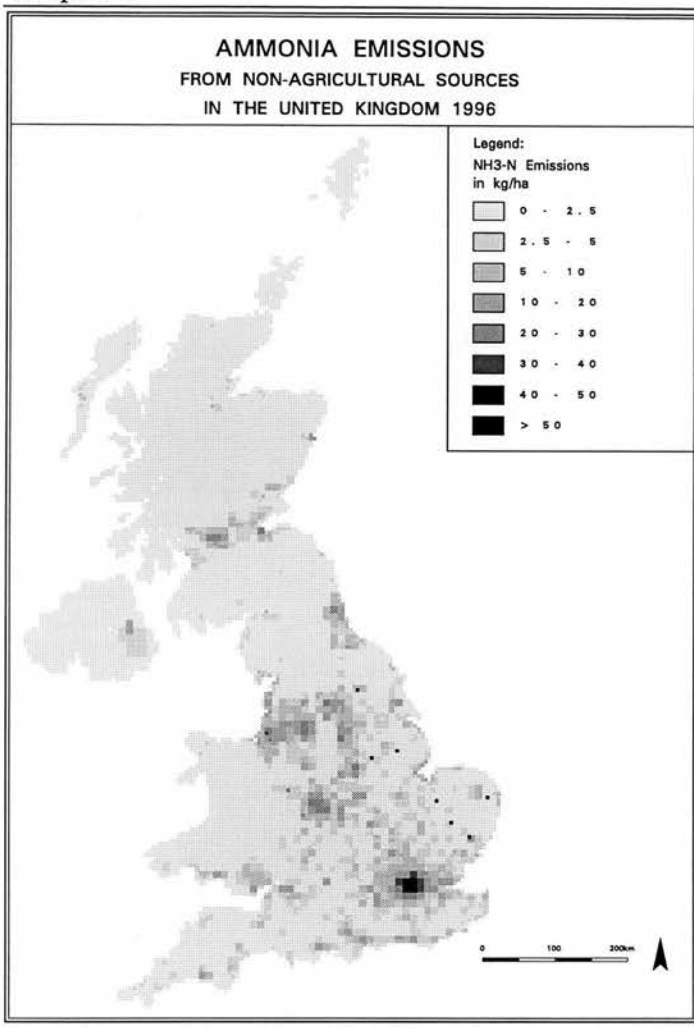


Figure 6.4. Ammonia emissions from non-agricultural sources in the UK for 1988/1996.

For the present study, the population distribution map (Figure 5.6.) was available at a 10 km resolution only, which provides a much coarser picture of the resulting NH_3 emissions scaled by population (Figure 6.4.). In Northern Ireland, the resolution of the population data and consequently the emissions linked to them is at a county level, i.e. even less spatially resolved. The other miscellaneous NH_3 sources, which were spatially distributed according to the landcover classes on which they are most likely to occur, were mapped at the 1 km level. Subsequently they were aggregated to 5 km grid cells, which were then combined with the population-based sources at a 10 km spatial resolution. The resulting map (Figure 6.4.) therefore presents the spatial distribution of emissions from these two groups of miscellaneous sources at two different resolutions, depending on the methodology used.

Only about 10% of all grid squares in the model for miscellaneous sources are estimated to emit more than 5 kg ha^{-1} NH_3-N, with nearly 80% of all squares emitting less than 2.5 kg ha^{-1} NH_3-N (Table 6.4., Figure 6.4.). Compared with the spatial distribution of agricultural NH_3 emissions, the miscellaneous sources combined under the heading of 'non-agricultural' emissions are mostly associated with areas with none or very little agricultural emission activity. Thus the two maps of agricultural and non-agricultural emissions complement each other for large parts of the UK (compare Figures 6.4. and 6.5.).

Table 6.4. Analysis of NH_3 emissions from non-agricultural sources in Great Britain for 1988/1996: % of 5 km grid squares in each category (classification as in Figure 6.4.).

Ammonia emission category (kg ha^{-1} NH_3-N)	proportion of GB grid squares
>0-2.5	78.6%
>2.5-5	11.4%
>5-10	6.4%
>10-20	2.6%
>20-30	0.6%
>30-40	0.2%
>40-50	0.1%
> 50	0.1%

Only the upland and hill areas show very low NH_3 emission levels on both the agricultural and the non-agricultural emission maps. Other noticeable exceptions to this apparent complementary distribution pattern are sewage spreading, biomass burning and a proportion of the wild animals, which are spatially concurrent with agricultural emissions. These sources together contribute, however, only a relatively small proportion of the total emissions on agricultural land.

6.5. SPATIAL DISTRIBUTION OF TOTAL AMMONIA EMISSIONS

The estimated total magnitude of NH_3 emissions from agricultural and other miscellaneous sources for the UK in 1988 amounts to 295 kt NH_3-N (see Table 6.1). The spatial pattern of agricultural NH_3 emissions shown in Figure 6.5. largely reflects the spatial distribution of livestock sources, especially cattle (Figure 6.2a) However, intensive arable farming areas in south-east England (see Figure 6.3.) are clearly visible in the combined map of all agricultural sources (Figure 6.5).

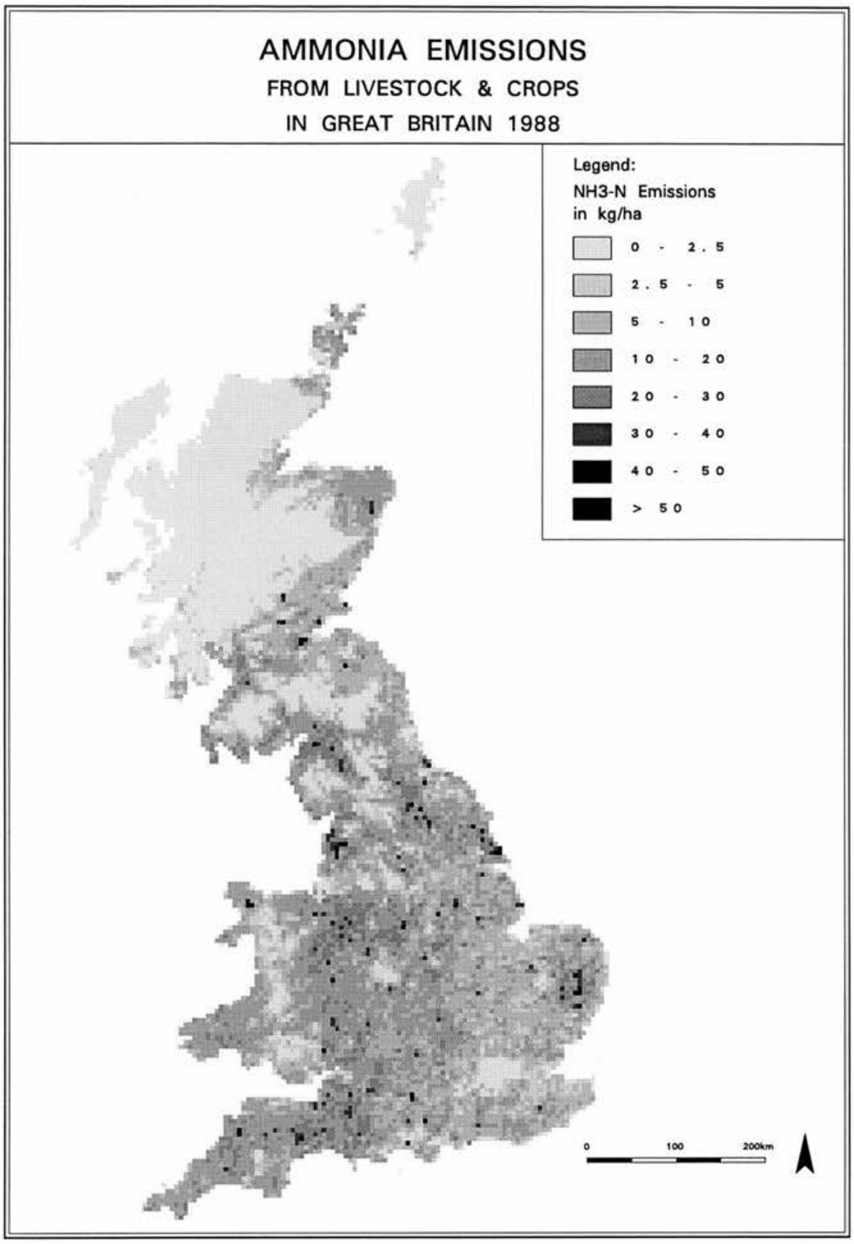


Figure 6.5. Total agricultural NH_3 emissions 1988 for Great Britain at a 5 km resolution (new model).

When combining the emissions from all agricultural sources and the non-agricultural emissions into one map (Figure 6.6.), the distinctive 'holes' in the agricultural emission map around London, in the Midlands, around Newcastle and in the Scottish Lowlands disappear. Everywhere else, i.e. in the non-urban areas, combining the non-agricultural emissions with the agricultural emissions does not have a substantial effect on the overall magnitude of emissions.

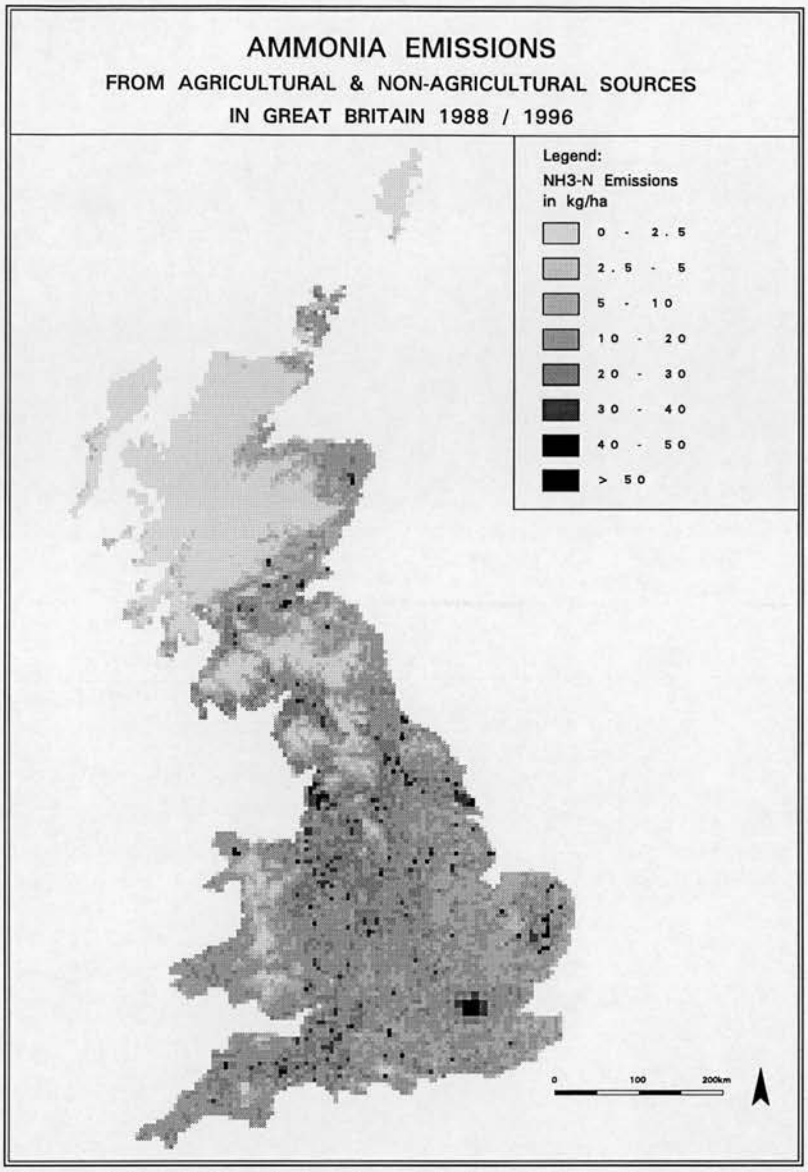


Figure 6.6. Total NH_3 emissions from agriculture (1988) and non-agricultural NH_3 emissions (1988/1996) for Great Britain at a 5 km resolution.

6.6. ANALYSIS OF THE SPATIAL INVENTORY FOR EMISSION CONTRIBUTIONS FROM AMMONIA SOURCE SECTORS

One of the main objectives of spatially distributed NH_3 emission inventories is the evaluation of the resulting maps for abatement potential. In order to meet this objective, it is essential that the importance of the different source categories is investigated. In any given grid square, emissions from livestock, fertiliser application or non-agricultural sources may contribute the majority of emissions. These main categories can be split further into contributions from, for instance, cattle, sheep, pigs and poultry.

Figures 6.7a and 6.7b show the contributions from agricultural and non-agricultural sources to the total emissions in 1988, respectively. The spatial distribution of the relative importance of these two main source groups is summarised quantitatively in Table 6.5. for all 5 km gridsquares in Great Britain.

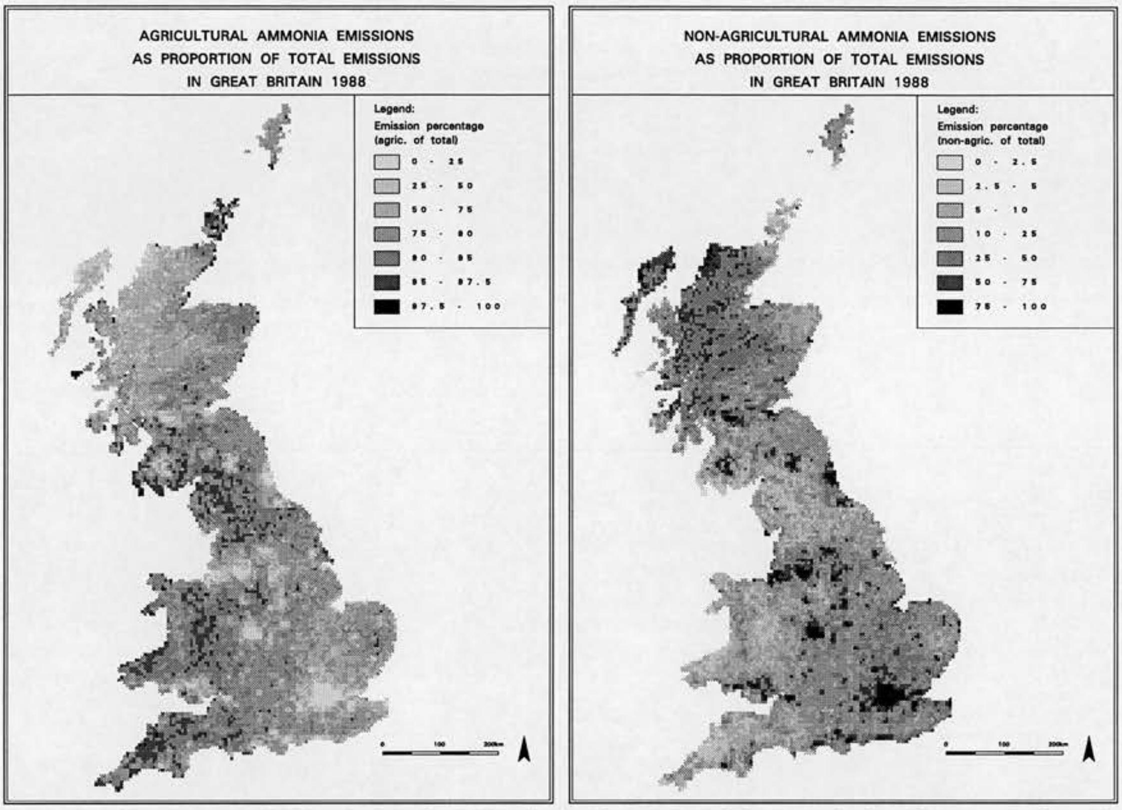


Figure 6.7. Contribution of NH_3 emissions from a) agricultural sources and b) non-agricultural sources to the total emissions in Great Britain in 1988.

Table 6.5. Proportion of GB grid squares with % contribution of NH_3 emissions from agricultural and non-agricultural sources to the total emissions and of NH_3 emissions from livestock and fertiliser application to the total agricultural emissions in 1988.

Category (% contribution to total or agric. Emissions)	Non-agric. Sources (Proportion of UK squares)	Fertiliser sources (Proportion of UK squares)	Category (% contribution to total or agric. emissions)	Livestock sources (Proportion of UK squares)	Agric. sources (Proportion of UK squares)
0-2.5	0.9%	15.4%	0-25	0.6%	3.6%
2.5-5	7.4%	10.7%	25-50	3.5%	8.9%
5-10	20.5%	28.2%	50-75	14.6%	23.6%
10-25	35.1%	27.2%	75-90	27.2%	35.1%
25-50	23.6%	14.6%	90-95	28.2%	20.5%
50-75	8.9%	3.5%	95-97.5	10.7%	7.4%
75-100	3.6%	0.5%	97.7-100	15.3%	0.9%

This analysis shows that emissions from agricultural sources provide more than half of the total emissions in 88% of 5 km grid squares, and more than three quarters of all emissions in 64% of all grid squares. This confirms the dominance of agricultural emissions over most of Great Britain.

Conversely, non-agricultural emissions contribute over 50% of the total emissions in only 12% of all grid squares (Figure 6.7b, Table 6.5.). Despite this low figure, a large proportion of grid squares (79%) show contributions of 5-25% of the total emissions from sources other than agriculture. This is especially apparent in the Highlands of Scotland, where large parts of agricultural land are very extensively used and total emissions are very low. Therefore, emissions linked to the human population and wild animals play a larger relative role in this and similarly structured areas.

Agricultural emissions are dominated (>50% contribution) by livestock emissions in 96% of all grid squares (Table 6.5.). The highest contributions from livestock emissions occur in the upland and hill areas of Scotland and northern England as well as on the Northern and Western Isles, where nearly all agricultural emissions are due to livestock farming (Figure 6.8a).

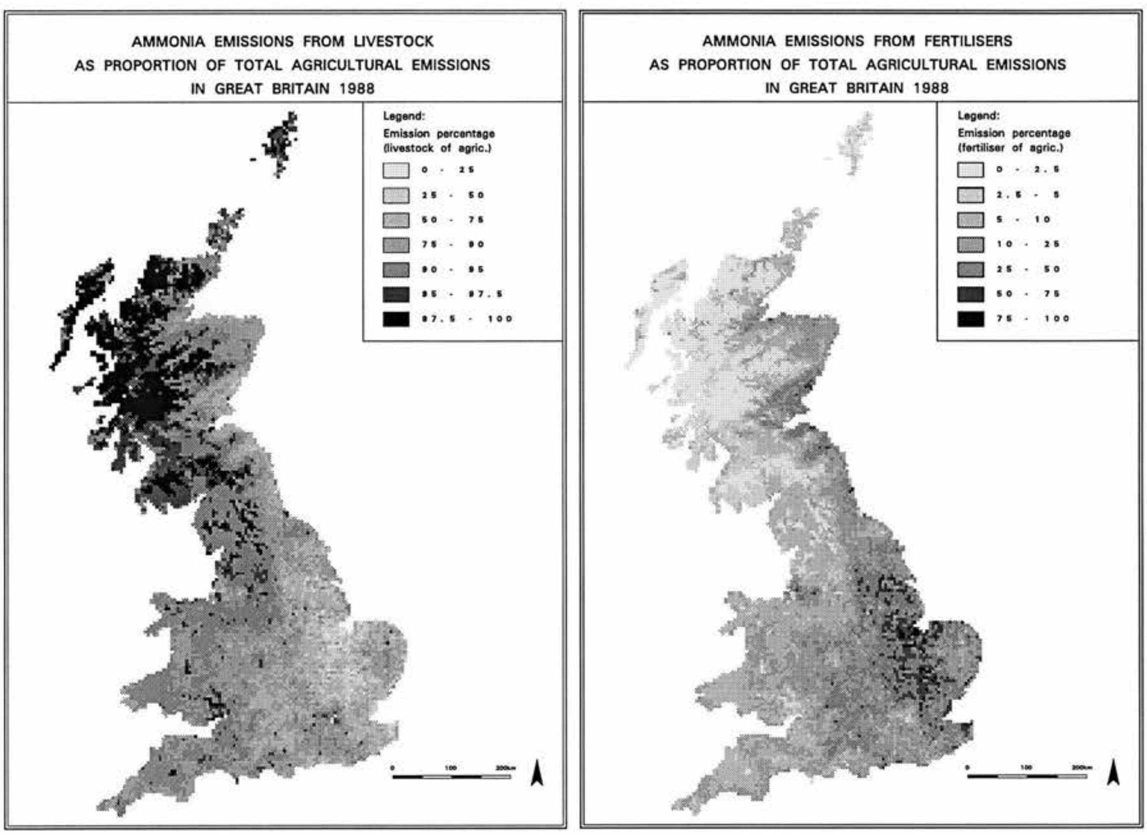


Figure 6.8. Contribution of NH_3 emissions from a) livestock sources and b) fertiliser application to crops and conserved grassland to the total agricultural emissions in 1988.

There is, however, a substantial proportion of grid squares, where emissions from fertiliser application to crops and conserved grassland play a significant role. Fertiliser emissions contribute over 25% of the total agricultural emissions in nearly 20% of all grid squares. These squares are situated in a band along the east coast of Britain from Aberdeen to Kent, which widens in East Anglia, where the highest values are found (Figure 6.8b).

Emission contributions from agricultural livestock were investigated further, as they provide the highest concentrations of NH_3 emissions and also the largest contribution to the total NH_3 emissions for the majority of Great Britain. For this purpose, the total livestock emissions were split into four sub-sources of cattle, sheep, pigs and poultry. Figures 6.9a-d show the magnitude of emissions from the 4 main livestock categories for all 5 km gridsquares in Great Britain in 1988 (see also Figures 6.2a-d).

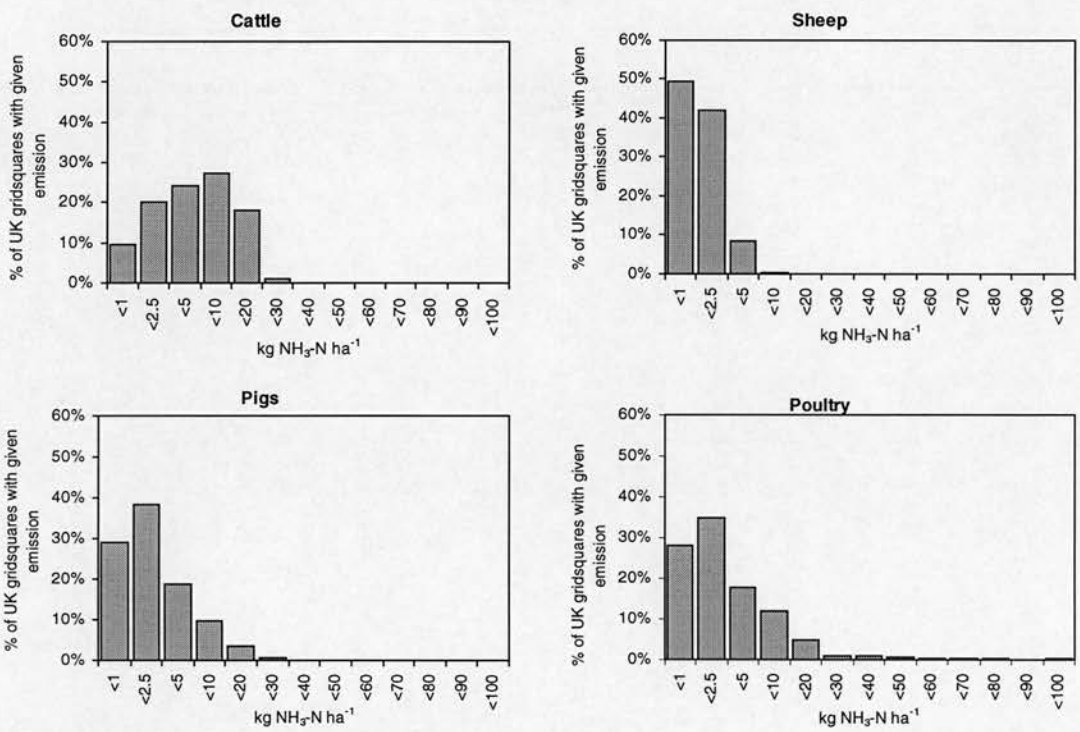
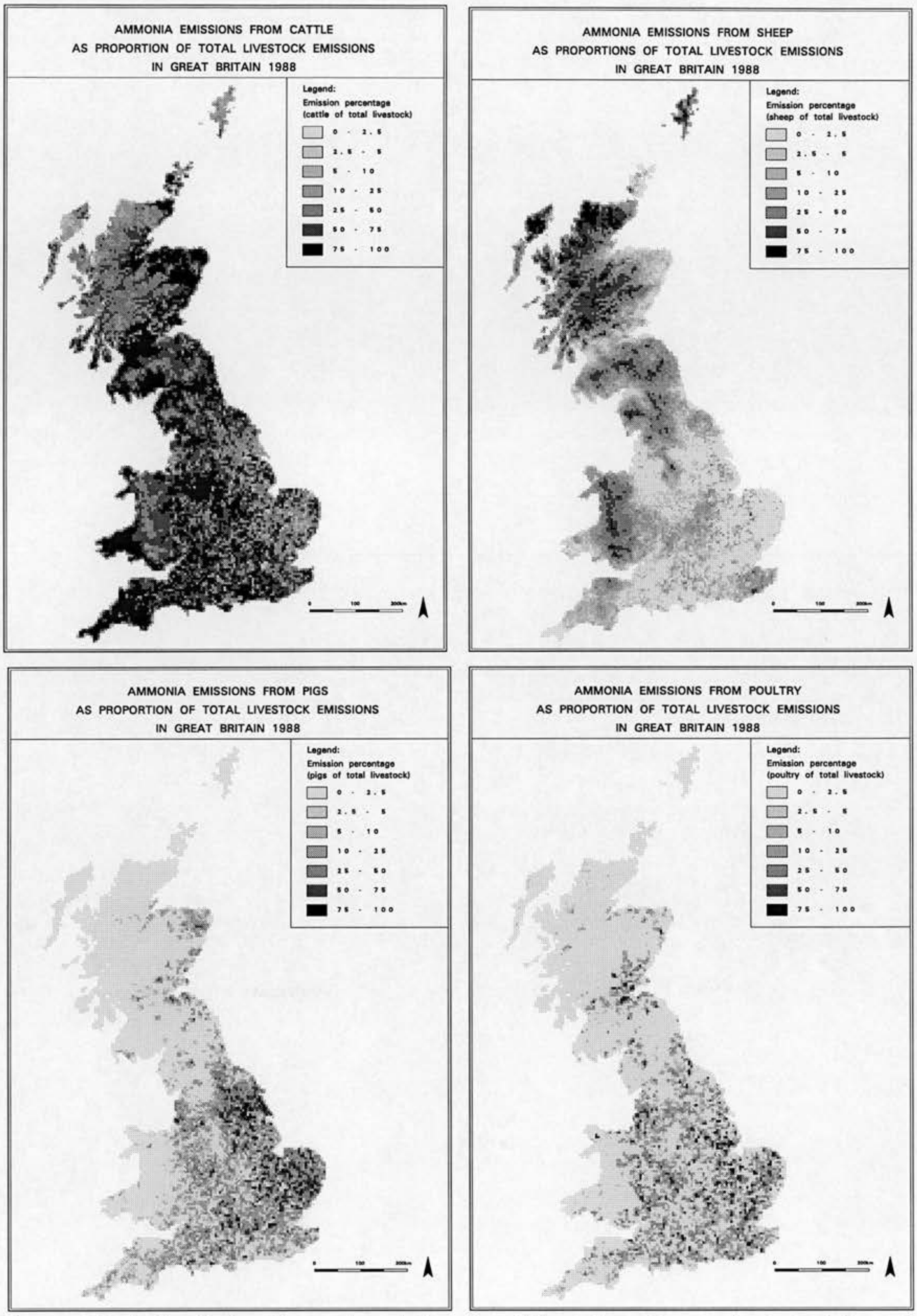


Figure 6.9a-d. Magnitude of ammonia emissions from different livestock categories, expressed as percentage of all 5 km gridsquares in Great Britain in 1988: a) cattle, b) sheep, c) pigs, and d) poultry.

Figures 6.10a-d show the relative contributions from the main livestock categories to the total livestock emissions in Great Britain in 1988. The spatial distribution pattern is distinctive for each type of livestock and is analysed further in the following paragraphs.



Figures 6.10a-d. Contribution of ammonia emissions from different livestock categories to total livestock emissions in Great Britain in 1988: a) cattle, b) sheep, c) pigs, and d) poultry.

Table 6.6. summarises Figures 6.10a-d by quantifying the relative contributions of the livestock categories within each 5 km gridsquare. Cattle dominate (>50% contribution) livestock emissions in 61% of all grid squares, as was expected due to their importance as the largest overall emission source (see Chapter 3 and Table 6.1.). Emissions from cattle contribute up to 30 kg N ha^{-1} Britain, with most 5 km squares in the range of 2.5-20 kg N ha^{-1}(Figure 6.9a). They provide the major source of agricultural NH_3 emissions in most lowland areas in Great Britain (Figure 6.10a).

Table 6.6. Proportion of GB grid squares classified for relative contributions from livestock categories to the total livestock emissions in 1988.

% contribution to livestock emissions	Cattle (Proportion of GB squares)	Sheep (Proportion of GB squares)	Pigs (Proportion of GB squares)	Poultry (Proportion of GB squares)
0-2.5	0.9%	25.5%	56.5%	65.1%
2.5-5	1.6%	14.4%	8.0%	7.2%
5-10	2.7%	16.1%	8.7%	7.5%
10-25	10.9%	18.4%	13.1%	10.7%
25-50	22.9%	13.3%	8.4%	6.4%
50-75	30.2%	9.3%	3.9%	2.3%
75-100	30.7%	2.9%	1.3%	0.9%

Sheep, on the other hand, are relatively more important as NH_3 sources in the upland and hill areas, where they provide large proportions of the total livestock and hence total NH_3 emissions (Figure 6.10b). Despite being present in most UK gridsquares, sheep contribute over half of the total livestock emissions in only 12% of all GB grid squares, and more than a quarter in only 26% of all squares (Table 6.6.). This is because the overall magnitude of emissions from sheep in any gridsquare is estimated to be relatively small, compared with other livestock categories, with sheep emissions exceeding 5 kg ha^{-1} in only very few squares (Figures 6.9b, 6.2b). These results show the relatively minor importance of sheep as an NH_3 source in the present inventory, compared with other emission sources, due to their very low emission density.

Emissions from pigs provide significant contributions to the total livestock emissions in a much smaller proportion of grid squares than sheep and cattle. In nearly two thirds of all grid squares, emissions from pigs contribute less than 5% of the livestock emissions. Only 5% and 1% of all grid squares, respectively, show values of over 50% and 75% livestock emission contributions (Table 6.6.). This is because pigs are generally farmed at a much higher intensity than cattle or sheep, and because intensive pig farming occurs mainly in a relatively confined area in the eastern, central and south-eastern parts of Britain (Figure 6.10c).

Poultry farming is even more intensive than pig farming and provides the highest emissions of all livestock categories, with estimates of up to 100 kg ha^{-1} on average in some 5 km gridsquares (Figure 6.9d). Emissions from poultry are relatively insignificant (<5% contribution) in 72% of all grid squares (Table 6.6.). Poultry provide over half of the total livestock emissions in only 3% of all grid squares, and over three quarters in less than 1% of all squares. On the map, poultry emission contributions are characterised by 'hot spots', rather than continuous areas with high concentrations (Figure 6.10d).

It is helpful to summarise the issues discussed above by showing the dominant emission source(s) in each gridsquare. An analysis was performed to show where a particular source sector contributes more than 45% of the total emissions in a square. The cut-off value of 45% was chosen as most suitable for this purpose, as 50% would have resulted in many squares not being assigned a dominant source category. Squares with total emissions of less than 1 kg ha^{-1} were assigned to a 'background' category and not analysed further. In any gridsquare where more than 1 category contributed over 45%, the category with the larger contribution was assigned as the dominant class. This occurred in only 8 gridsquares. Pigs and poultry, both intensive and non-land based agricultural sources, were combined into one category for this analysis to show the overall pattern more clearly. The results are mapped in Figure 6.11. Figure 6.12. shows the frequency distribution of the total NH_3 emissions for the dominant source types mapped in Figure 6.11.

Most of the Scottish Highlands and Islands are characterised by very low emission estimates (< 1 kg ha^{-1}), which were classified as background. Outside this area, the map highlights again the dominance of cattle for most of Great Britain (Figure 6.11.). Sheep provide the most important emission source in only very few squares. These are located in upland and hill areas in Wales, north-east England and the Scottish Borders (Figure 6.11.). The squares dominated by sheep emissions are characterised by generally low total emission estimates (see Figure 6.12b).

Intensive pig and poultry farming is dominant mainly in eastern lowland areas of Great Britain, where it contributes to the largest total emissions per grid square. Of all squares with emissions > 40 kg N ha^{-1} $year^{-1}$, 56% are dominated by pigs and/or poultry, 27% by non-agricultural emissions, 10% by mixed sources and only 7 % by cattle. This low frequency of cattle dominated squares with high emissions confirms the less intensive nature of cattle farming, compared with pig and poultry farming, as far as NH_3 emissions are concerned. Emissions from fertiliser application to crops and conserved grassland are a major feature in East Anglia, however, the grid squares dominated by this source provide generally small emission totals of up to 10 kg ha^{-1}.

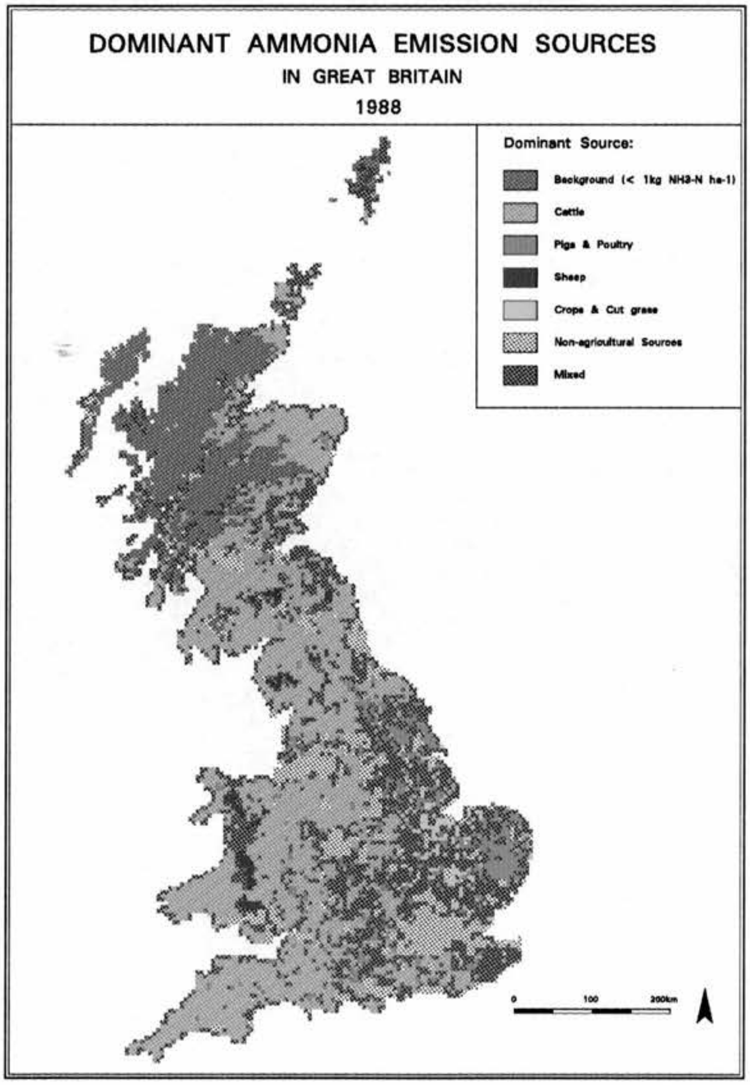


Figure 6.11. Classification of 5 km gridsquares in Great Britain according to estimated dominant emission source category in 1988. Squares with < 1kg N ha^{-1} are referred to as background. Squares with >45% contribution from a given category are referred to as dominated by that source.

A large proportion of squares dominated by non-agricultural sources is found around major population centres. Figure 6.12e indicates that this category provides the dominant source in some squares with very large emission totals, similar to pigs and poultry, but to a lesser degree. Future efforts to model the spatial distribution of these sources more accurately, involving e.g. newly available data for the location of industrial sources, may shift these sources away from the population centres they are associated with in the present model. There are also large areas where no single source category provides > 45% of the total NH_3 emissions.

The results in Figures 6.11. and 6.12. are dependent on the accuracy of the input data to the underlying emissions model, e.g. emission source strength data and spatial redistribution rules. For instance, if the emission source strength estimates for a source type are underestimated in the model, as is likely for sheep (see Section 3.2.2.), the pattern of dominant source types may be changed significantly with improved source strength estimates. Another example is the relatively crude spatial distribution methodology for non-agricultural sources in the present model, which is assumed to overestimate emissions in population centres, and thus biasing Figures 6.11. and 6.12. to some extent. However, despite these uncertainties, Figure 6.11. shows clearly the main spatial distribution pattern of the different dominant sources.

Figure 6.12. demonstrates the different frequency structure of the emissions from the main source sectors, highlighting the importance of pigs and poultry as well as some non-agricultural sources as the sectors that are most likely to cause extreme adverse effects to the environment. Thus pig and poultry emissions have a larger local impact than other emission source categories for squares where they are present in large numbers. Cattle and sheep emission sources, on the other hand, are more evenly distributed than the more localised intensive poultry or pig sources, thus causing less acute impacts (see Section 1.2.) in their immediate neighbourhood.

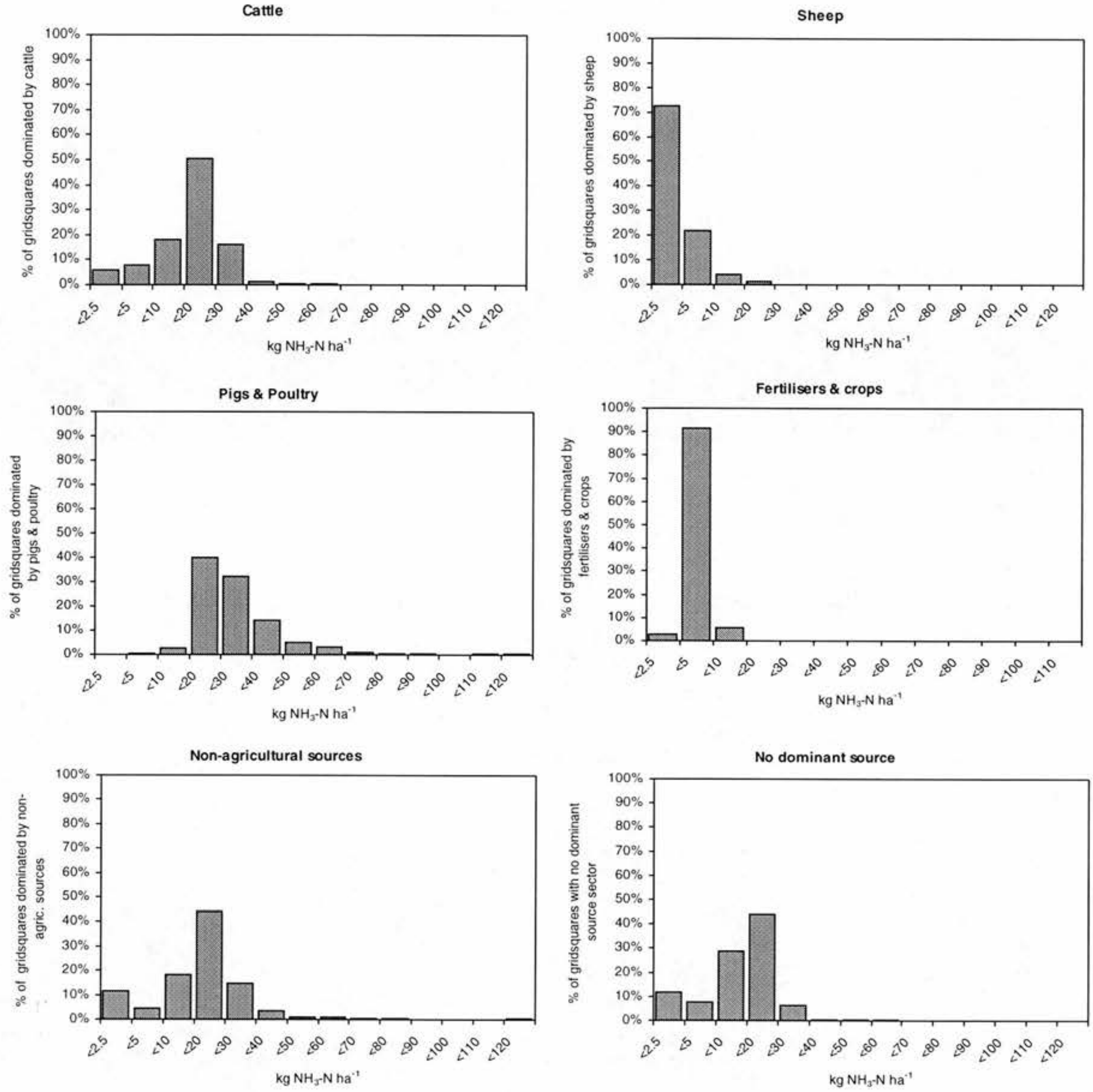


Figure 6.12. Dominant emission sources in GB 1988: Frequency distribution of NH_3 emissions (kg N ha^{-1}) in gridsquares dominated (>45%of emissions) by a) cattle, b) sheep, c) pigs and poultry, d) fertilisers and crops, e) non-agricultural sources and f) with no distinctive dominant source.

Summarising Figures 6.11. and 6.12., the following can be stated:

- Areas dominated by sheep and fertilisers are in general linked with low total NH_3 emissions (1-5 kg N ha^{-1} in a 5 km square).
- Areas dominated by cattle or with no distinctive dominant source are mostly linked with emissions in the medium range (20-30 kg N ha^{-1} in a 5 km square).
- Areas dominated by pigs and poultry and some urban areas generally show estimated NH_3 emissions at the higher end of the range of total emissions per 5 km gridsquare (10-120 kg N ha^{-1}).

6.7. THE SPATIAL PATTERN OF AMMONIA EMISSIONS IN NORTHERN IRELAND

The present study includes Northern Ireland in a spatially distributed emissions inventory at a fine resolution for the first time. The results are therefore described separately here.

Total NH_3 emissions from agricultural sources in Northern Ireland in 1996 are estimated at 24.9 kt NH_3-N or 30.3 kt NH_3, with a contribution of 23.7 kt NH_3-N (95% of total) from livestock farming and 1.2 kt NH_3-N (5% of total) from fertiliser application to crops and (cut) grassland (Tables 6.7., 6.8.). The emissions from livestock originate mainly from cattle farming, which contributes 73% of the total agricultural emissions. Cattle farming is even more important as an NH_3 source in Northern Ireland than in the rest of the UK, where only 55% of all agricultural emissions are from cattle. Other major NH_3 sources in Northern Ireland are poultry farming (2.8 kt NH_3-N), pig farming (1.7 kt NH_3-N) and sheep farming (0.9 kt NH_3-N).

Most NH_3 sources are located in the lowland areas north, south and west of Loch Neagh, along the rivers Mourne and Foyle towards the western border with the Republic of Ireland, around Strangford Lough in the southeast and Upper and Lower Lough Erne (Figure 6.13.-6.15.). The less fertile areas of the Antrim, Mourne, Armagh and Sperrin Mountains represent the more extensively used agricultural areas with less emissions.

Table 6.7. Northern Ireland: Summary of Agricultural Census statistics and NH_3 emission estimates for 1996 (emission source strength estimates from DoE, 1995).

Category	Animal numbers	Emission animal^{-1} (kg NH_3-N year^{-1})	Total NH_3-N (kt year^{-1})	Total NH_3 (kt year^{-1})	Contribution (%)
Cattle	1,629,085	11.23	18.29	22.21	73.5
Sheep	2,445,964	0.38	0.93	1.13	3.7
Pigs	541,330	3.18	1.72	2.09	6.9
Poultry	14,645,404	0.19	2.78	3.38	11.2
Total livestock	-	-	23.73	28.81	95.2
Total crops & grass	-	-	1.21	1.46	4.8
Total	-	-	24.93	30.28	100.0

The maps of NH_3 emissions shown in Figures 6.13.-6.15. are compatible with the new emission maps for Great Britain with regard to time (June Censuses 1996 for

Great Britain and Northern Ireland), their spatial resolution (5 km by 5 km grid squares) and the NH_3 source strength estimates, which are based on the official NH_3 emission estimates (DoE, 1995). Therefore, the maps were subsequently combined to form a map covering the entire United Kingdom (Figures 7.1.-7.3.), which is also relevant for the modelling of atmospheric transport and deposition.

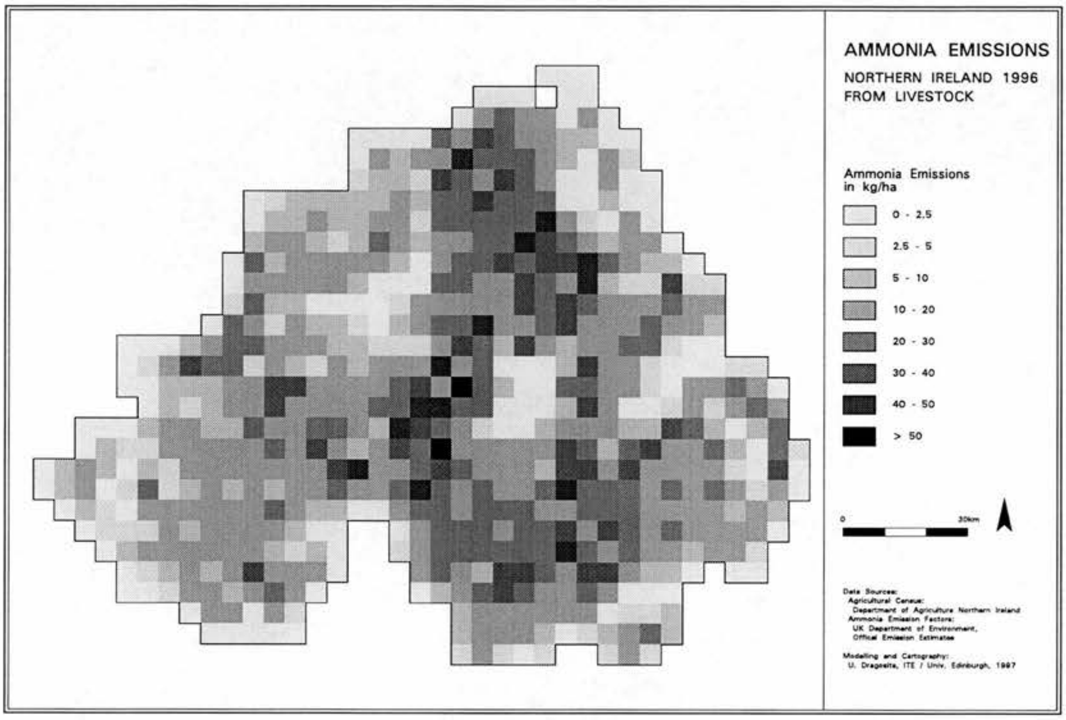


Figure 6.13. Northern Ireland: NH_3 emissions from livestock 1996.

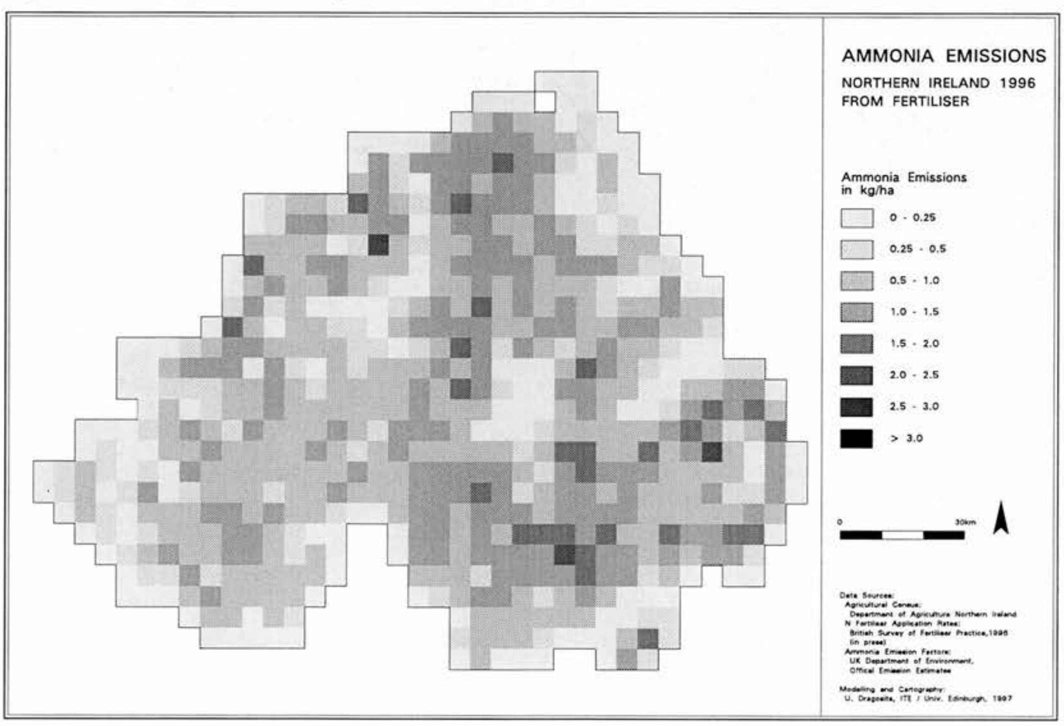


Figure 6.14. Northern Ireland: NH_3 emissions from fertilised crops and grassland 1996.

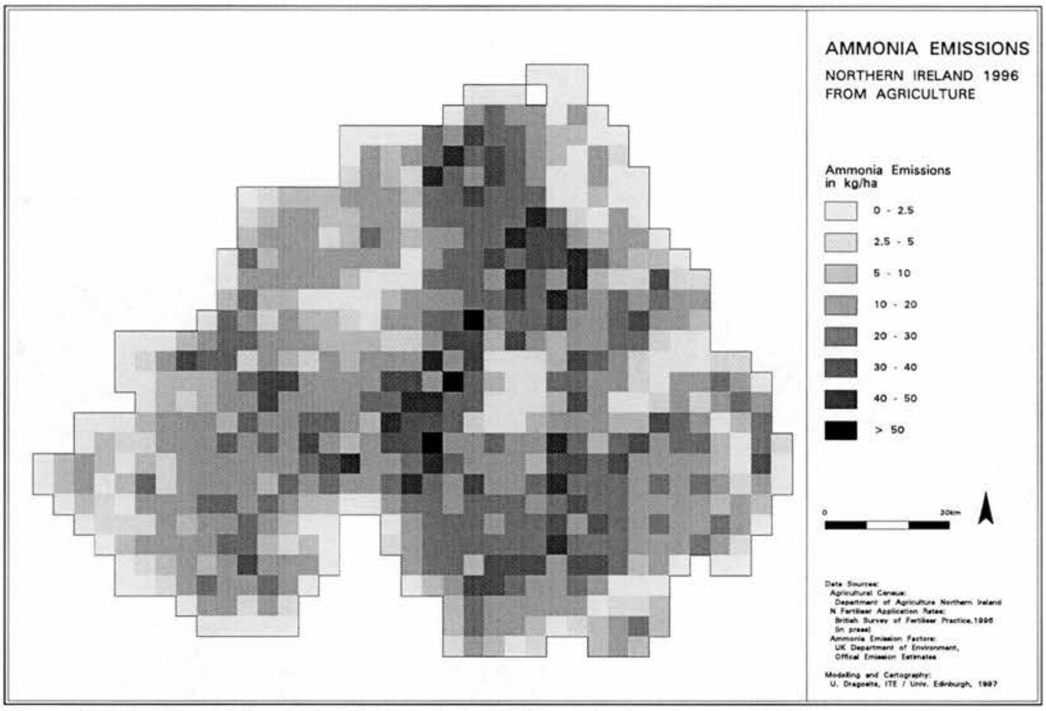


Figure 6.15. Northern Ireland: Total NH_3 emissions from agricultural sources 1996.

Although the contribution of emissions from Northern Ireland in the UK context is only approximately 11% (see Table 6.8.), its location south-west of southern and western Scotland is expected to be reflected in atmospheric transport and deposition models as (locally) increased deposition to the areas downwind of the sources.

Table 6.8. Contribution of agricultural NH_3 emissions of Northern Ireland to total UK emissions in 1996 (numbers may not add up to 100% due to rounding).

Sectors	UK (kt NH_3-N)	Great Britain (kt NH_3-N)	Northern Ireland (kt NH_3-N)	Northern Ireland (% of UK)
Cattle	133.7	115.4	18.3	13.7
Sheep, goats & deer	15.9	14.9	0.9	5.9
Pigs	23.9	22.2	1.7	7.2
Poultry	27.8	25.1	2.8	10.0
Horses	2.0	2.0	-	-
Total livestock	203.2	179.5	23.7	11.7
Fertiliser	29.7	28.5	1.2	4.1
Total	232.9	208.0	24.9	10.7

6.8. DISCUSSION AND CONCLUSIONS

This chapter has presented and evaluated the results of applying the new methodology to model the spatial distribution of NH_3 emissions at the national level (5 km grid), which was developed as part of this thesis. Main issues covered in detail are the comparison of the new methodology with previous efforts, as well as the analysis of contributions by different source types and the implications of their spatial distribution.

The application of the new NH_3 source distribution model described in Chapters 4 and 5 resulted in a more realistic spatial pattern of NH_3 emissions for the UK than that achieved by previous, more general models. A comparison of the results of this study with earlier efforts to model NH_3 emissions shows two main differences: the first is due to the different level of spatial detail and the way NH_3 emission sources are distributed over the country. The second is due to differences in the NH_3 source strength data applied in the models.

For instance, Kruse (1986) and Kruse *et al.* (1989) used agricultural census data provided at a 10 grid resolution for the base year of 1981 (Figure 1.7.). Their maps cover only England and Wales, and only include agricultural NH_3 sources. Overall, the total emissions resulting from their model are much larger than the results presented here. This is mainly due to larger emission source strength estimates, especially for cattle (19.3 kg N animal^{-1} year^{-1}) and sheep (2.68 kg N animal^{-1} year^{-1}). Their emission estimate from fertiliser applications, however, is much lower, due to a smaller volatilisation factor of 1% applied in their model. It should also be noted that the redistribution methodology for NH_3 sources such as agricultural livestock is less critical for model results at a 10 km grid level than at a 5 km resolution, due to the averaging effect of the coarser resolution.

Eager (1992) used agricultural census data from 1988 at a 5 km grid resolution for the first time. These data were provided by the Edinburgh University Data Library, following Hotson's (1988) redistribution model. Eager was the first to map NH_3 emissions for Scotland at a fine resolution (5 km), but did not combine the separate maps for England & Wales and Scotland into a single map for Great Britain. The emission factors used in his model for sheep and cattle are again much larger than in the present study.

Sutton *et al.* (1995) combined Eager's two separate maps into a single map for Great Britain for 1988 and included spatially distributed NH_3 emissions from non-agricultural sources for the first time. Dragosits *et al.* (1996b) applied the 'official' emission factors (DoE, 1995) to the same agricultural census data and also included non-agricultural emissions (from Sutton *et al.*, 1995).

In this study, emission sources such as agricultural livestock, fertiliser applied to crops and conserved grassland were distributed specifically as NH_3 sub-sources i.e. livestock grazing, housing, manure storage and land-spreading, rather than equally distributed over all agriculturally utilised land. This resulted in much smaller and more realistic emission estimates for extensively grazed upland and hill pastures, while emissions were concentrated in more intensively used lowland areas. Thus the new national inventory accomplished an improved representation of the typical pattern of NH_3 emissions, where source and sink areas are in close proximity to each other. This is important for modelling deposition and impacts due to the steep gradients of NH_3 deposition downwind of sources.

The main source types have been analysed here regarding their contributions to the total UK NH_3 emissions, as well as regarding their spatial pattern. The source categories investigated in detail were total livestock, fertiliser use on crops and conserved grassland and non-agricultural sources. It was shown that the agricultural and non-agricultural emission maps are complementary regarding the spatial distribution of emissions, with rural areas dominated by agricultural emissions. Emissions from livestock are largest in lowland areas with intensive agriculture, mainly in the western part of England, in Wales and in southern and eastern Scotland. The largest emissions from fertiliser use on crops and conserved grassland occur in the eastern counties of England.

Due to their relative importance, livestock emissions were divided further into contributions from cattle, sheep, pigs and poultry. Each source type shows a distinctive pattern: pigs and poultry provide the most intensive sources per unit area with distinctive 'hot spots' on the maps, whereas sheep provide large proportions of the relatively low total emissions of upland and hill areas. Cattle are shown to be the dominant NH_3 source for large parts of lowland Britain, especially in the west.

The combined inventory of NH_3 emissions from both agricultural and non-agricultural sources was analysed further to determine the dominant source type for each square of the national 5 km map. Despite some uncertainties, e.g. regarding the spatial distribution of non-agricultural sources, a distinctive pattern was shown: squares where sheep or fertiliser emissions provide the largest contribution are relatively scarce, due to the low emissions per unit area of sheep and arable farming land, compared with other sources. These squares show generally low emission rates of 1-5 kg N ha^{-1}. Cattle dominate the magnitude of emissions in a large number of squares, with average emissions in the range of 5-30 kg N ha^{-1}. Where intensive, non-land based livestock farming (pigs and poultry) and non-agricultural sources dominate the total emissions, the magnitude is generally highest, with emissions ranging from 10-120 kg N ha^{-1}.

It should be noted that such a map of dominant emission sources (Figure 6.11.) is naturally dependent on the accuracy of the underlying source strength data for each sector. Thus if the much larger emission estimates of Kruse *et al.* (1989) had been applied, this would be reflected in Figure 6.11. with a larger area dominated by sheep emissions. While accepting these uncertainties, it is clear that Figure 6.11. nevertheless summarises the major regional pattern of source sector dominance.

Chapter 7

Temporal changes in modelled spatial patterns of ammonia emissions over the UK

7.1 INTRODUCTION

Over the last decades, British agriculture has changed significantly. Consequently the number of NH_3 emission sources, the emission source strength estimates and thus the spatial patterns of NH_3 emissions have altered. In previous chapters the model results were discussed in detail, regarding the spatial pattern as well as the absolute and relative importance of individual source sectors and their specific characteristics. Chapters 4-6 have so far focused on results for 1988, to facilitate the comparisons undertaken, and to ensure none of the disclosivity rules imposed on the 1996 census data were broken by presenting detailed output.

In this chapter, the main aim is to provide a spatio-temporal view of NH_3 emissions in Great Britain, for the period from 1969 to 1996. The choice of the time period was determined by the availability of spatially distributed Agricultural Census data. The best model input data were available for 1988 and 1996, regarding agricultural census, landcover and emission source strength data. For both years it was possible to apply the new redistribution approach developed in this thesis. For 1969 and 1988, spatially redistributed agricultural census data (using Hotson's (1988) model) were available at a 5 km grid resolution from the Edinburgh University Data Library.

It should be noted that changes in the Agricultural Census over time make exact comparisons difficult. Examples for this are changes in the questions asked in the June Agricultural Census, changes in the census items recorded, as well as changes in the threshold farm sizes for farms required to fill in the forms. For this reason, and because of restrictions due to the different source redistribution approaches used for 1969 and 1996, it was decided to perform the comparison in two steps (1969-1988, 1988-1996), starting with the more recent period.

Potential future changes in the magnitude of NH_3 emissions in the UK, due to the implementation of abatement measures are also considered. Most potential abatement measures are by nature source-specific, in that specific measures may be devised for different livestock categories or for fertilisers and crops. The effects of implementing abatement measures are thus spatially distributed, according to the spatial location of the different source categories (Section 7.4.), rather than spread equally over the country.

7.2. TEMPORAL CHANGES BETWEEN 1988 AND 1996

Total NH_3 emission estimates for the United Kingdom appear to have declined very slightly between 1988 and 1996, as shown in Table 6.1. The causes of this change are not readily identifiable as the fine details of the totals in this table are not directly comparable, mainly due to changes in the Agricultural Census. For instance, poultry emissions for 1988 in the table do not include turkey numbers for England and Wales, because of changes in the Census questionnaires between the two years.

As far as emission from livestock are concerned, again the total magnitude stayed largely unchanged between 1988 and 1996, despite a decrease in emissions from pig farming and an increase in emissions from poultry farming. The latter is largely due to the inclusion of turkeys. Cattle numbers declined steadily from 1990 to 1995, but increased sharply in 1996 (GSS, 1997). This increase may be temporary and could be linked to a rise of cattle numbers on farms due to the crisis in the cattle market caused by BSE. As a result of this farmers were not able to sell their cattle. The contribution of cattle to emissions for the 1996 Agricultural Census numbers is estimated to be approx. 2 kt of NH_3-N higher than in 1995, an increase of 1.5%. Emissions from the application of N fertiliser to crops and grassland have decreased by approximately 3 kt NH_3-N, due to a downward trend in fertiliser application rates over the last few years.

Although the changes in total NH_3 emissions are rather small, changes in the spatial distribution of emissions are more evident (see Figures 7.1.-7.4. and Tables 7.1.-7.2.). Any differences between 1988 and 1996 discussed in the following paragraphs do not include Northern Ireland, as no spatial data were available for 1988 for this study. It should be noted that the new redistribution methodology was used for the compilation for all maps for 1996 and 1988 in this section.

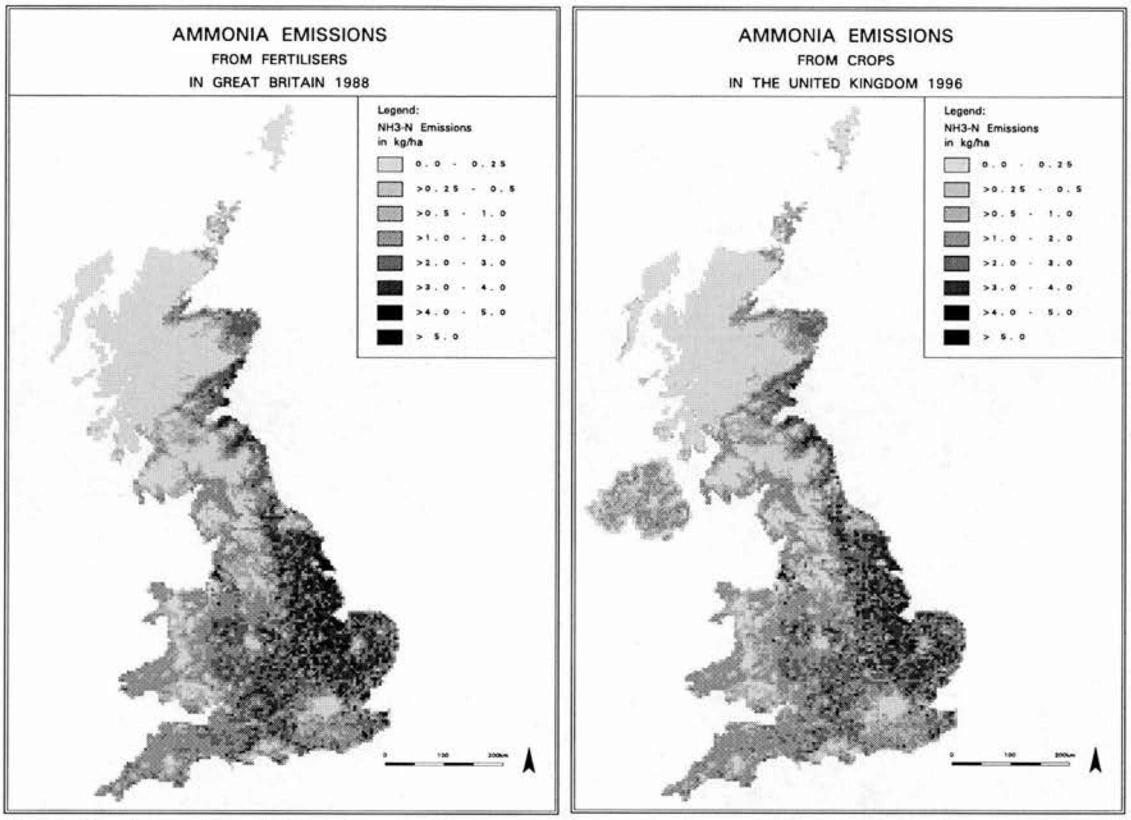


Figure 7.1. Ammonia emissions from fertiliser application to crops and conserved grassland a) Great Britain 1988, b) United Kingdom 1996 (NB: both maps were created using the new redistribution methodology).

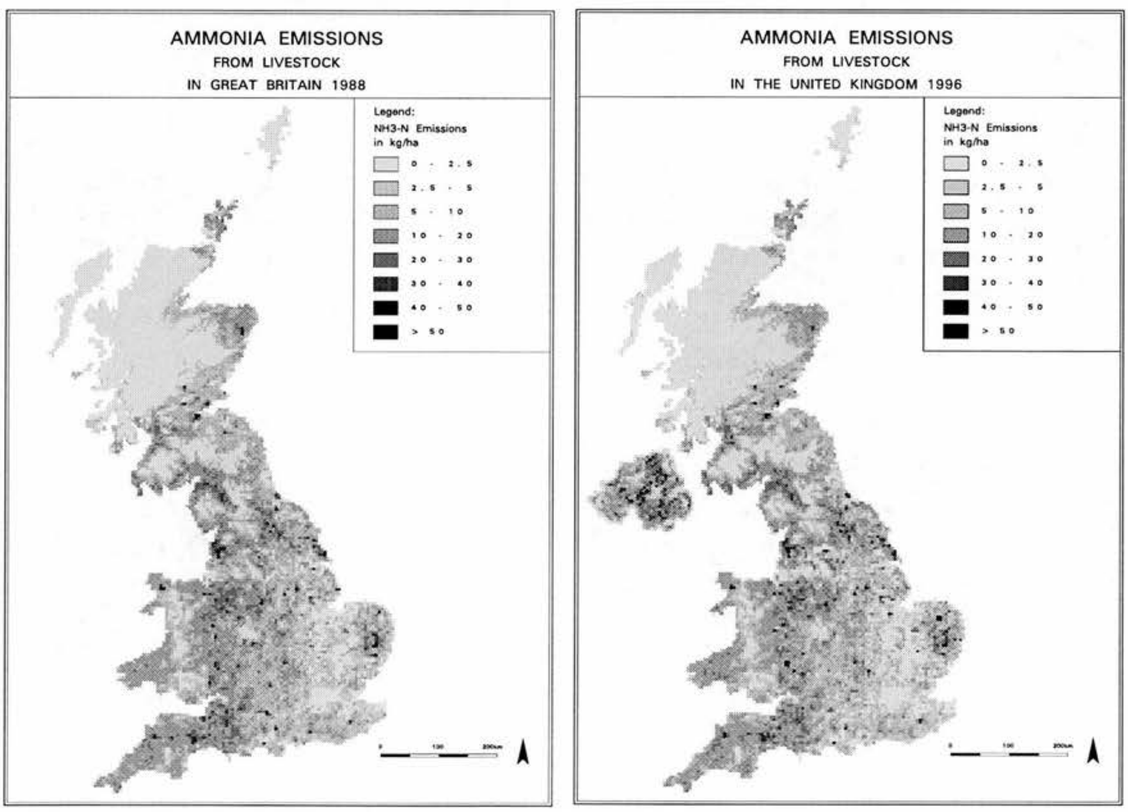


Figure 7.2. Ammonia emissions from livestock a) Great Britain 1988, b) United Kingdom 1996 (NB: both maps were created using the new redistribution methodology).

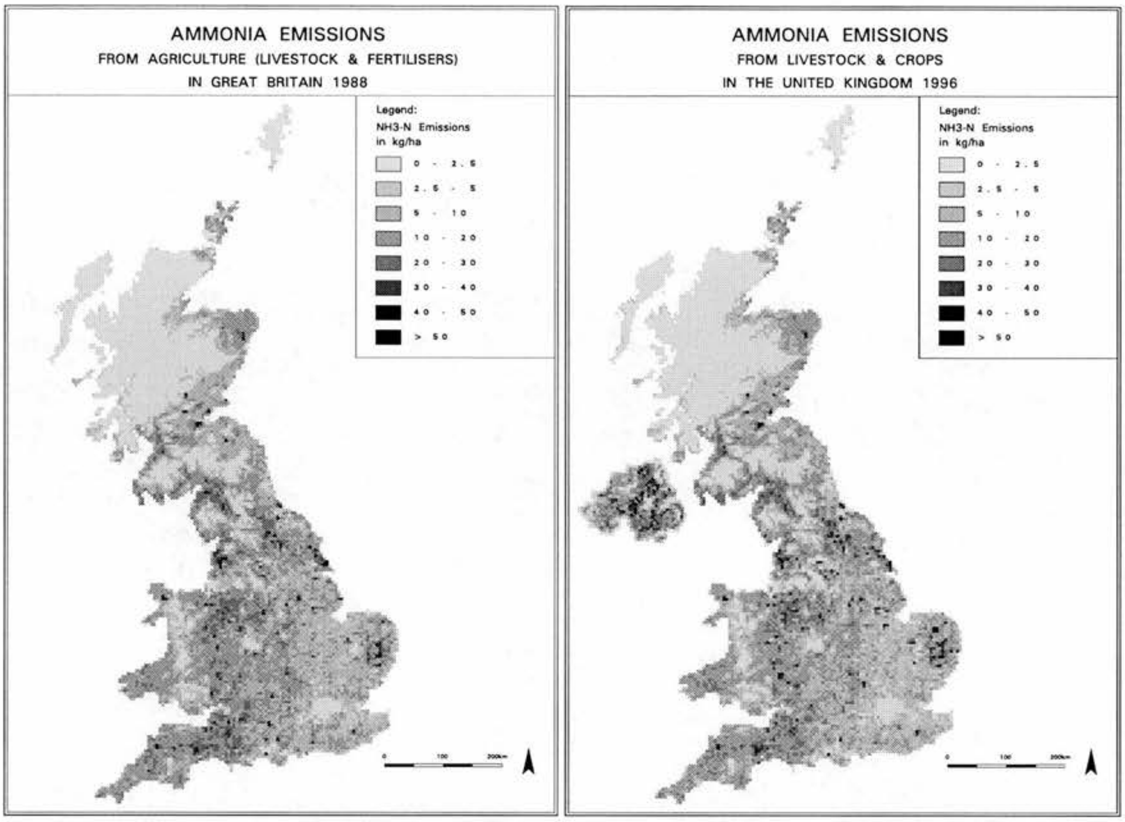


Figure 7.3. Ammonia emissions from all agricultural sources a) Great Britain 1988, b) United Kingdom 1996 (NB: both maps were created using the new redistribution methodology).

The absolute differences between the 2 reference years were compared for each gridcell on a 1 km and 5 km model output level. An analysis of the 1 km model results is also included here to give an indication of the variability of NH_3 emissions at a finer resolution than the 5 km grid squares chosen in this study. For most areas, differences in total agricultural NH_3 emissions between 1988 and 1996 are quite small (Table 7.1.; Figures 7.3., 7.4a, 7.5.), with about a quarter of all 1 km and 5 km grid squares with less than ± 0.1 kg N ha^{-1} difference, and about 60% of all 1 km and 5 km grid squares in 1996 within a range ± 1 kg N ha^{-1} of the 1988 estimates.

The higher variability at the 1 km level is significantly smoothed by the aggregation to the 5 km level. For the 5 km grid results, the range is -60 to +80 kg ha^{-1} in the squares with the largest difference between the two years. For the 1 km grid results, this range is -500 to +1700 kg ha^{-1}. There is a significant number of grid squares with very large changes at both the 1 km and the 5 km resolution. These squares mostly represent areas with intensive livestock farming, the positive changes showing new developments, intensification or relocation since 1988, the negative changes representing disappearance, extensification or relocation. However, some of these changes may be artefacts due to differences in the two datasets: As mentioned above (Section 4.2.1.), the census data for 1988 were provided as non-disclosive parish data, with any disclosive parishes amalgamated for each county in England and Wales. Although the livestock emissions from the county-summary parishes provide only a small proportion of the total livestock emissions in England and Wales in 1988 (1.4% for cattle, 1.1% sheep, 2.3% pigs, 0.6% poultry in 1988), localised effects may still be considerable. For instance, the summary parishes of 5 counties contain between 19,000 and 24,000 pigs each, and there are 6 county-summary parishes with 50,000 – 130,000 birds.

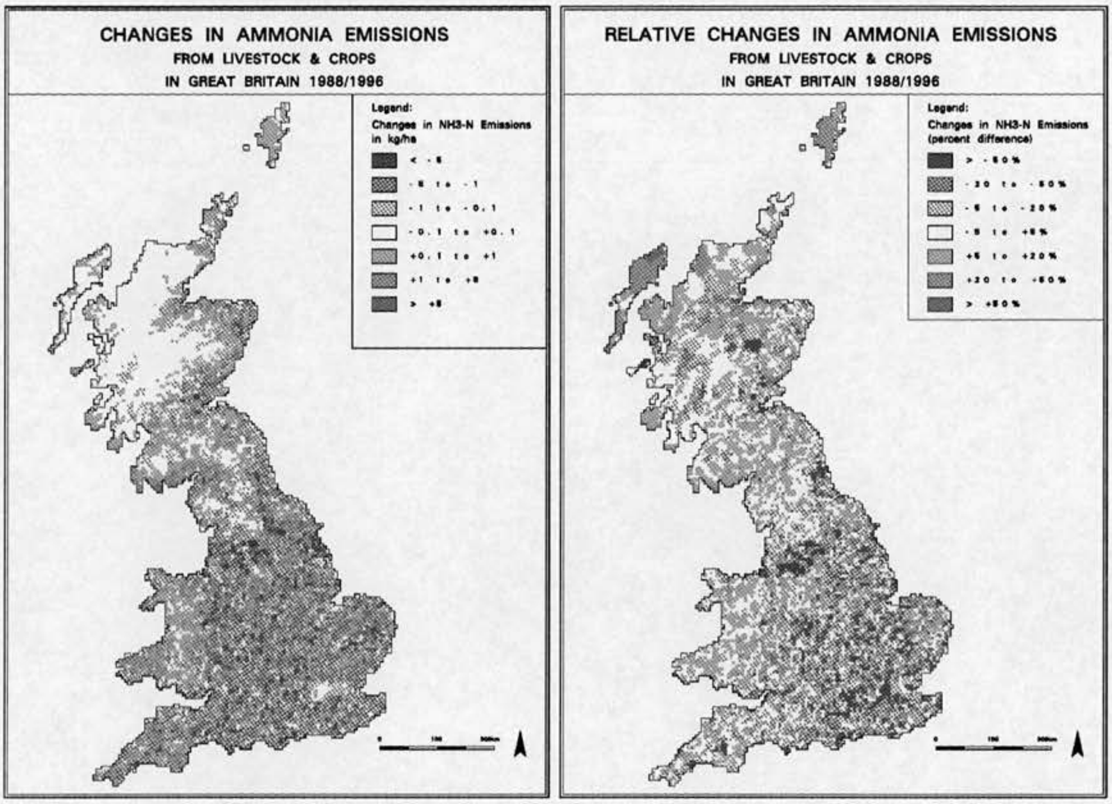


Figure 7.4. Changes in NH_3 emissions from agricultural sources in Great Britain 1988 to 1996 a) absolute changes (in kg N ha^{-1}) and b) relative changes (in %); NB: these maps are based on the new redistribution methodology). Positive values indicate increased emissions between 1988 and 1996, negative values indicate decreases.

For 1996, it was possible to assign parishes, which were contributing to disclosive output grid squares, to adjacent parishes as described previously. The consequences of choosing either the county-summary approach or the new approach described above were investigated further for 1996, by implementing both methods and comparing the results. The differences in the emissions inventory due to these 2 approaches are much larger for 1996 than for 1988, because the disclosivity threshold for parish data was raised from 3 to 5 holdings per parish. The results of the sensitivity analysis carried out for 1996 are described in detail in Section 9.2.1.

Table 7.1. Differences between 1988 and 1996 for spatially distributed inventories of agricultural emissions in Great Britain; positive values indicate increased emissions from 1988 to 1996, negative values indicate decreased emissions.

Difference (1996-1988) ($kg\ NH_3\text{-}N\ ha^{-1}$)	Area (km^2) 5 km grid	% of total area (5 km grid)	Area (km^2) 1 km grid	% of total area (1 km grid)
50 to 1700	225	0.09	334	0.2
30 to 50	100	0.04	434	0.2
20 to 30	250	0.1	712	0.3
10 to 20	1,500	0.6	2,686	1.2
5 to 10	3,800	1.5	6,557	2.8
1 to 5	38,275	15.2	34,544	15.0
0.1 to 1	53,550	20.2	36,683	15.9
0.1 to -0.1	58,500	23.2	61,120	26.5
-0.1 to –1	43,600	17.3	34,196	14.8
-1 to –5	45,250	17.9	39,955	17.3
-5 to –10	5,825	2.3	8,716	3.8
-10 to –20	1,375	0.5	3,478	1.5
-20 to –30	150	0.06	762	0.3
-30 to –50	125	0.05	388	0.2
-50 to –500	25	0.01	205	0.09

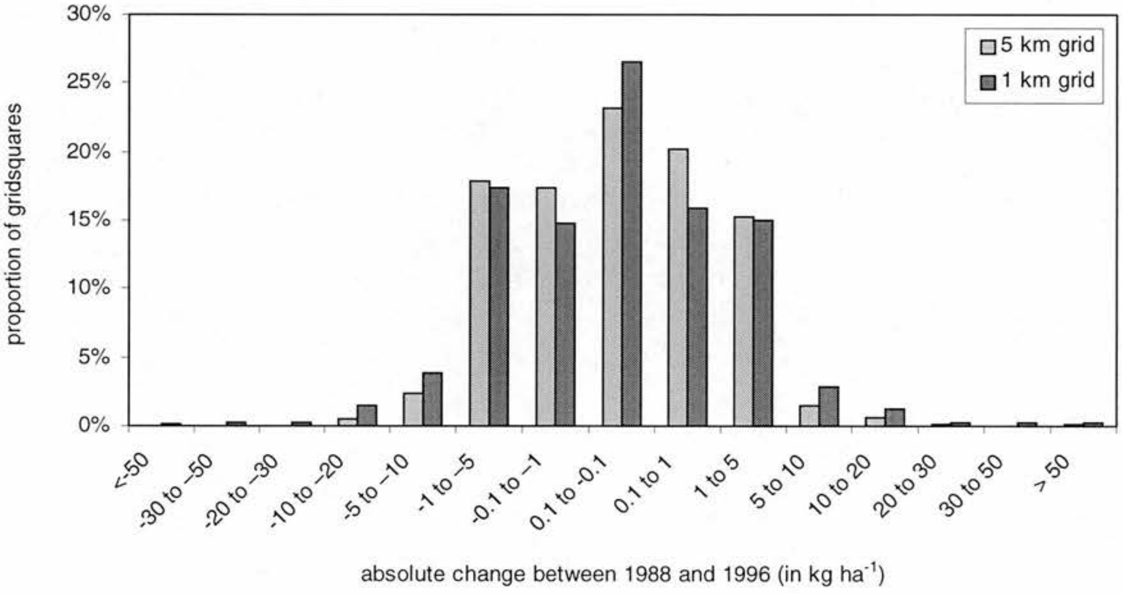


Figure 7.5. Absolute change in NH_3 emissions (in kg ha^{-1}) between 1988 and 1996 in Great Britain at a 1 km and 5 km grid resolution. Positive values indicate increased emissions from 1988 to 1996, negative values indicate decreased emissions.

The spatial pattern of absolute change for Great Britain shows several significant trends (Figure 7.4a, 7.5.). The larger positive and negative changes (>± 5 kg ha^{-1} in a 5 km square) only occur in a small number of grid squares and appear to be linked particularly to pig and poultry farming. Smaller increases between 1988 and 1996 (0.1-5 kg ha^{-1} in a 5 km grid square), showing some intensification, occur more frequently. These increases can be linked to the more fertile agricultural areas. Smaller decreases (-0.1 to -5 kg ha^{-1} in a 5 km grid square) appear to have happened mainly in the same areas, possibly showing some structural changes at the local level. These would merit closer investigations, if access to farm level data for both years were available.

Hardly any change (expressed in absolute terms; as kg N ha^{-1}) seems to have occurred in marginal areas such as large areas of the Highlands and Islands of Scotland, the higher areas of the Scottish Borders, Pennines, Welsh Hills and large urban conglomerations. A few obvious explanations for these patterns can be found in the agricultural census data. The main reasons for the apparently increasing agricultural activities in Wales (causing some smaller increases in NH_3 emissions) appear to be a substantial increase in sheep numbers and a smaller increase in cattle. The lack of large intensive pig and poultry units in Wales explains the relative absence of large increases or decreases for this area. Cattle and pig emissions appear to have increased in some areas, but decreased in other areas. Pig emissions have increased in Norfolk, North Yorkshire and eastern Scotland, and decreased in Humberside and most of the rest of England. Cattle emissions have risen in Dumfries and Galloway, but decreased in eastern Scotland and most of England.

An analysis of relative changes in the magnitude of NH_3 emissions between 1988 and 1996 was also carried out, and the results are shown in Figures 7.4b and 7.6. as well as Table 7.2. Emission levels in the low emission areas have changed considerably in relative terms, although they may not have changed very much in absolute terms, compared with the high emission areas (Figure 7.4a). In some areas, e.g. the Highland & Islands of Scotland, increases or decreases by > ±50% are common, which indicate a similar rate of change as in the high emission areas in England. It should also be noted that the relative differences between the 2 years are smoothed out by the aggregation from the 1 km to the 5 km grid resolution.

Table 7.2. Relative differences between 1988 and 1996 for spatially distributed inventories of agricultural emissions in Great Britain: positive values indicate increased emissions from 1988 to 1996, negative values indicate decreased emissions.

Category	Area (km^2)	% of total area	Area (km^2)	% of total area
< -50%	3,175	1%	6,760	3%
-20 to –50%	22,500	9%	24,444	11%
-5 to –20%	66,025	26%	54,793	24%
-5 to +5%	67,300	27%	52,350	23%
+5 to +20%	49,575	20%	40,008	17%
+20 to +50%	28,150	11%	27,630	12%
>+50%	15,825	6%	24,785	11%

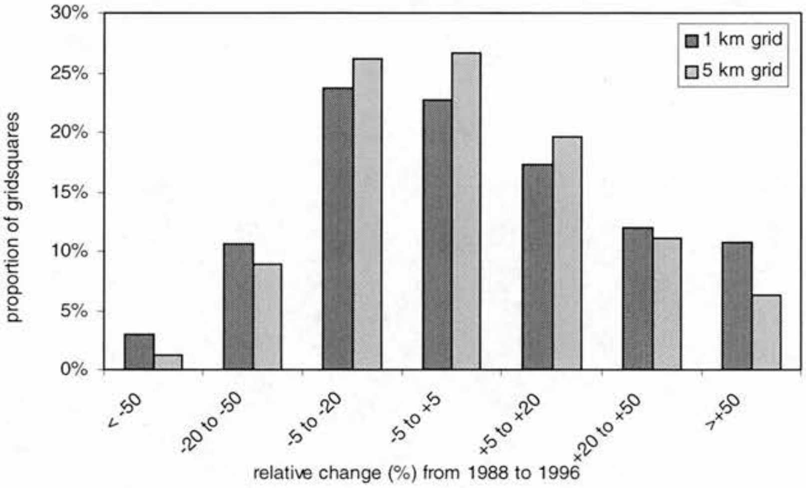


Figure 7.6. Relative changes in NH_3 emissions (in %) between 1988 and 1996 in Great Britain at a 1 km and 5 km grid resolution. Positive values indicate increased emissions from 1988 to 1996, negative values indicate decreased emissions.

7.3. TEMPORAL CHANGES BETWEEN 1969 AND 1988

It should be noted that, as for 1988 and 1996, the 1969 NH_3 emission totals are largely affected by the contribution from agricultural livestock. Changes in livestock numbers and distribution are therefore expected to be the main factors affecting changes in the pattern of the total NH_3 emissions between 1969 and 1988. Table 7.3. indicates that the largest changes in livestock numbers have occurred for sheep (+54%). This large increase can be related to changes in the sheep market, mainly due to increases in demand for lamb, e.g. in Spain and France, and the consequent rise in production (J. Dick, ITE Edinburgh, pers. comm., 1998). Regarding other livestock types, only relatively small changes in the total numbers have occurred. However, there have been substantial changes in the demographic structure of each of the livestock types. For example, there has been a significant decrease in the fraction of dairy cattle since 1969. This can be related to the introduction of milk quotas in 1984. Such changes in policy have affected average NH_3 emissions per average cattle animal because of the different N excretion rates from different cattle sub-classes (see Section 3.2.1.).

Table 7.3. Changes in livestock demography in the UK between 1969 and 1988. After GSS, 1972 and GSS, 1990.

Animal numbers	1969	1988	% difference
CATTLE			
Dairy cows & heifers in calf & milk	3,274,752	2,911,313	-11%
Beef cows & heifers in calf & milk	1,210,682	1,373,350	13%
Other cattle > 2 yrs.	1,767,738	1,615,410	-9%
Other cattle 1-2 yrs.	2,647,214	2,688,572	2%
Other cattle < 1yr.	3,473,737	3,283,476	-5%
Total cattle	12,374,123	11,872,121	-4%
SHEEP			
Adults	14,581,882	20,355,032	40%
Lambs (<1 yr.)	12,021,692	20,587,272	71%
Total sheep	26,603,574	40,942,304	54%
PIGS			
Breeding sows	914,698	804,423	-12%
Other pigs for breeding & fatteners	6,868,312	7,175,921	4%
Total pigs	7,783,010	7,980,344	3%
POULTRY (FOWLS)			
Layers	75,479,113	48,624,742	-36%
Breeders	6,714,521	6,878,821	2%
Table birds	38,417,744	75,305,083	96%
Total poultry	120,611,378	130,808,646	8%

There are, however, other - potentially even more important - factors to take into account. Intensification in British agriculture has lead to increased NH_3 emissions per animal. Although there were more cattle in the UK in 1969 than in 1988, with a higher proportion of dairy cows, smaller rates of fertiliser N input to grassland are estimated to result in much smaller total emissions, due to much lower emissions per animal.

Given the changes in the fertiliser application rates to grassland and general agricultural practice (regarding livestock feeding etc.), there is considerable uncertainty over the emission source strength factors to be applied for 1969. Fertiliser application rates to the main crops or crop groups are well documented by the Surveys of Fertiliser Practice for England & Wales and Scotland (separate surveys until 1992). For instance, average N application rates for grass were 118 kg ha^{-1} in 1996 in England and Wales (Burnhill *et al.*, 1997), compared with about 75 kg ha^{-1} in 1970 (Burnhill *et al.*, 1996). This would result in much smaller average cattle grazing emissions for 1969 than in 1988, according to the response curve developed by Pain *et al.* (1997), which describes the relationship between fertiliser input to grazed pasture and NH_3 emissions from cattle (see Section 3.2.1., Figure 3.4.). Assuming a similar relationship for conserved grass, which is used for winter feeding, an overall lower N excretion rate from cattle can be assumed for 1969. Consequently, total source strength estimates for cattle can be assumed to be smaller in 1969 than in 1988.

In order to adjust emission source strength estimates for the different livestock classes, more work is required to relate farming practice and feeding regimes to emission source strength estimates which reflect the situation in 1969. For the comparison between 1969 and 1988, two approaches were followed. Firstly, the same source strength figures that were used for 1988 were applied to all livestock classes in 1969, providing a high source strength scenario. Secondly, the 1988 emission source strength estimates were decreased experimentally by 20% for grazing livestock (cattle, sheep) for 1969, to provide a low source strength scenario (see Table 7.4.). The figure of 20% was selected on the basis of the updated N response curve of Pain *et al.* (1997), using the changed mean fertiliser rates for grassland for the different years. Although this is recognised as very uncertain, it is considered to be more likely than the upper estimate assuming no change, and more research is needed to improve these estimates. Both sets of emission source strength data were incorporated in a simple spatial model, which used already redistributed census data (as per Hotson's (1988) method). Thus, regarding the spatial distribution methodology, the results for 1969 (see Figures 7.7a, b) are compatible with the version of the 1988 emission inventory (Figure 7.7c), which was developed on the basis of the redistributed census data according to Hotson (1988).

Table 7.4. Comparison of NH_3 emissions from agricultural livestock for 1969 and 1988; high (as 1988) and low emission source strength scenario (-20% for grazing livestock) applied to 1969 livestock census data.

Livestock Category	Livestock numbers 1969	Livestock numbers 1988	Emissions 1969 low scenario (kt NH_3-N)	Emissions 1969 high scenario (kt NH_3-N)	Emissions 1988 (kt NH_3-N)
Cattle	12,374,123	11,872,121	111.2	139.0	133.3
Sheep	26,603,574	40,942,304	8.1	10.1	15.6
Pigs	7,783,010	7,980,344	24.8	24.8	25.5
Poultry	120,611,378	130,808,646	22.9	22.9	24.9
Total livestock	-	-	167.0	196.8	199.2

The spatial pattern of livestock emissions in Great Britain has changed substantially over the period of 1969 – 1988. The higher variability of the 1988 emissions can be associated with the increase of larger livestock farms as well as reduced activity on marginal land (see Figures 7.7a-c). This has resulted in an increased impact of local NH_3 emissions in areas with intensive sources, as was expected.

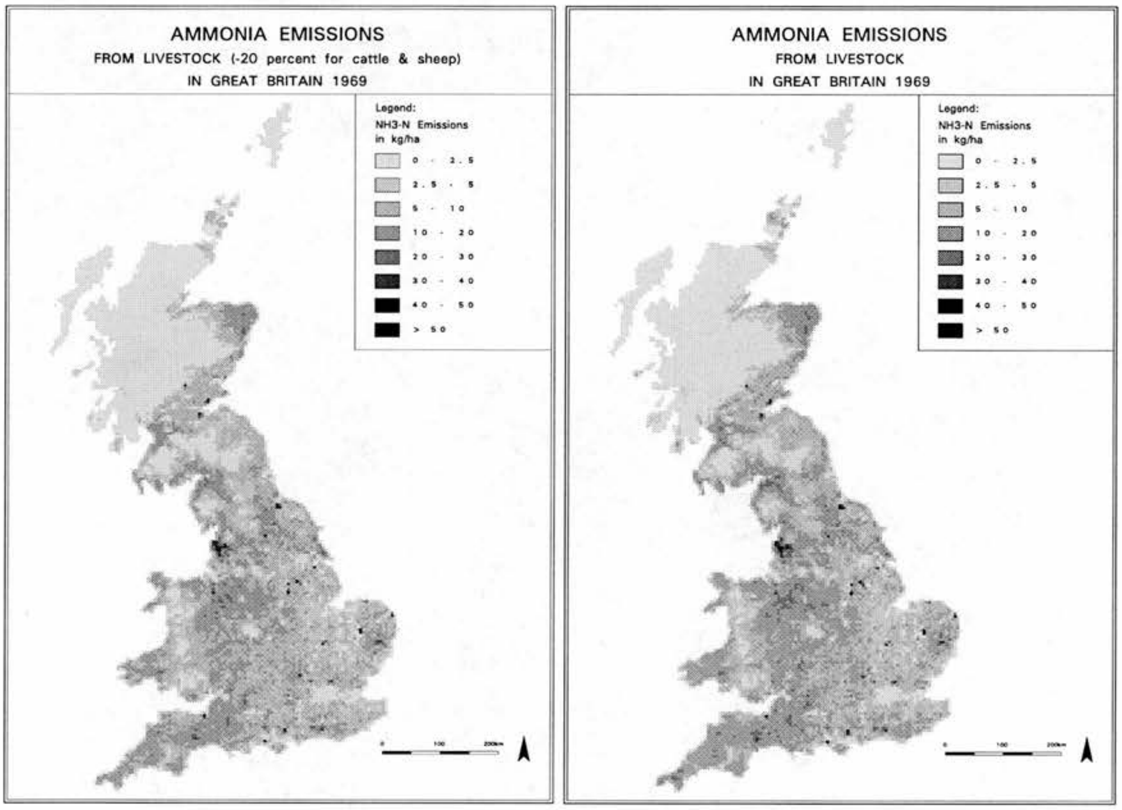


Figure 7.7. Ammonia emissions from livestock in Great Britain in a) 1969 (low emission scenario); b) 1969 (high emission scenario); (redistribution of Agricultural Census data for both 1969 and 1988 according to Hotson's (1988) methodology). (NB: emissions for 1969 and 1988 were mapped using the redistribution methodology of Hotson (1988) to ensure comparability).

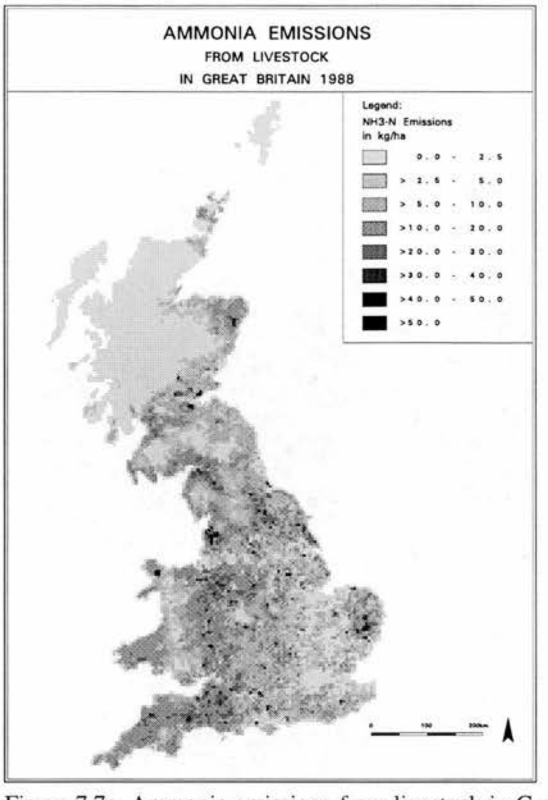


Figure 7.7c. Ammonia emissions from livestock in Great Britain in c) 1988. (NB: emissions for 1969 and 1988 were mapped using the redistribution methodology of Hotson (1988) to ensure comparability).

For crops and conserved grassland, fertiliser N inputs have generally increased between 1969 and 1988 (Chalmers *et al.*, 1989; Church, 1974; Eager, 1992). The areas under wheat and oilseed rape, both crops receiving high levels of N fertiliser have increased significantly between 1969 and 1988 (Table 7.5.). Total crop emissions for 1969 and 1988 are estimated at 17.3 kt and 31.3 kt for Great Britain, respectively. This constitutes a rise of 81 % between the two years. The effects of this intensification are clearly visible in Figures 7.8a and b.

Table 7.5. Comparison of areas of main crops and conserved grassland and their average fertiliser application rates for 1969 and 1988 (GSS, 1973 and 1993; Chalmers *et al.*, 1989, Church, 1974).

Crop category	Fertiliser application 1969 (kg N ha^{-1})	Fertiliser application 1988 (kg N ha^{-1})	area 1969 (ha)	area 1988 (ha)
Wheat	84	192	833,216	1,885,533
Barley	79	123	2,412,676	1,877,583
Oats	67	98	382,394	120,220
Total potatoes	162	194	248,428	179,847
Sugar beet	148	123	184,796	200,546
Oilseed rape	187	228	5,211	347,104
Ley grass	91[a]	115[a]	2,306,533	1,612,879
Permanent grass	91[a]	115[a]	4,997,073	5,160,988

[a] no distinction between different grassland categories, values for average grassland.

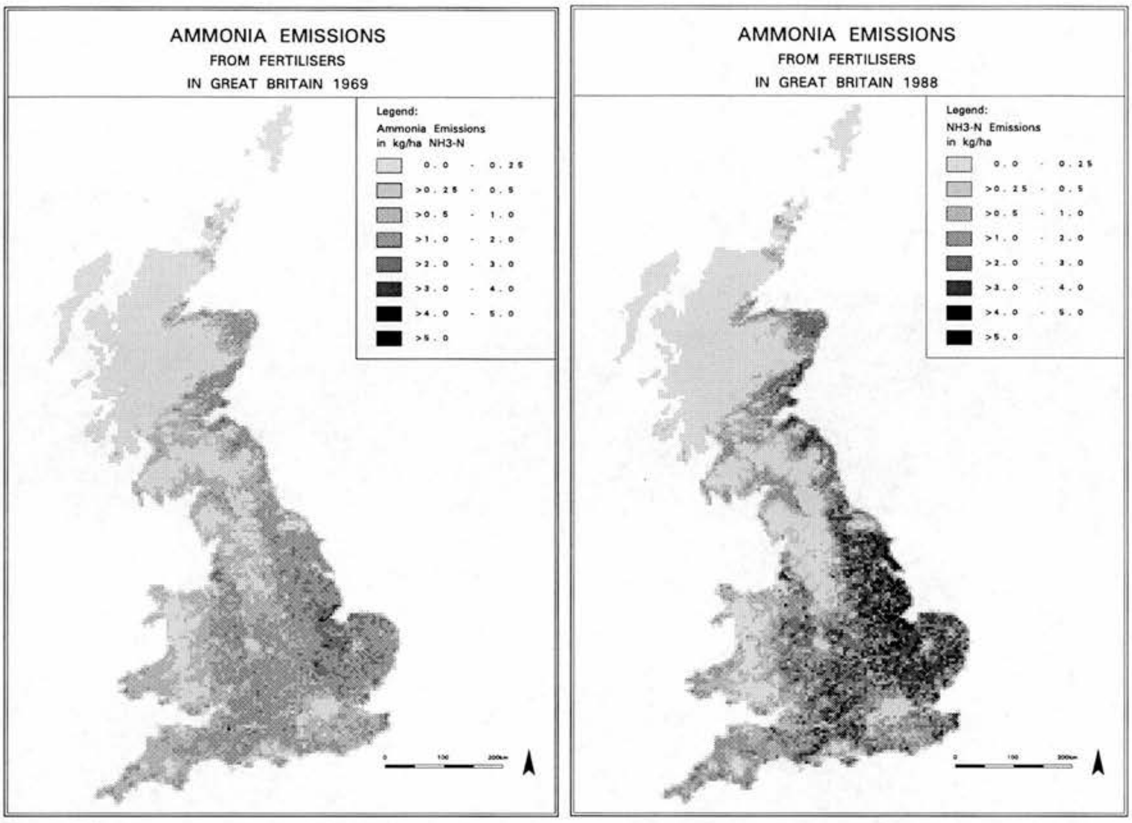


Figure 7.8. Ammonia emissions from crops and conserved grassland in Great Britain in a) 1969 and b) 1988 (NB: both scenarios for 1969 and 1988 were mapped using the redistribution methodology of Hotson, 1988).

A comparison of the total agricultural emissions was carried out between the two reference years. Figures 7.9. (a-c) show the different spatial patterns for the 2 scenarios for 1969 and the 1988 inventory, based on the spatial distribution of agricultural census data by Hotson (1988). The absolute and relative changes between 1969 and 1988 were analysed using a) the low emission scenario for 1969, with smaller source strength estimates for cattle and sheep, and b) the high emission scenario for 1969, with the same source strength estimates as for 1988. While the temporal changes calculated using the low scenario for 1969 are estimated to represent a more realistic picture (Figures 7.10a, 7.11a), it is difficult to separate the effects of changing spatial distribution patterns for livestock and crops from the effects of estimated changes in source strength data. In order to investigate the influence of the different spatial distributions of NH_3 sources in isolation, absolute and relative changes were also mapped using the same source strength data for the reference years (Figures 7.10b, 7.1b).

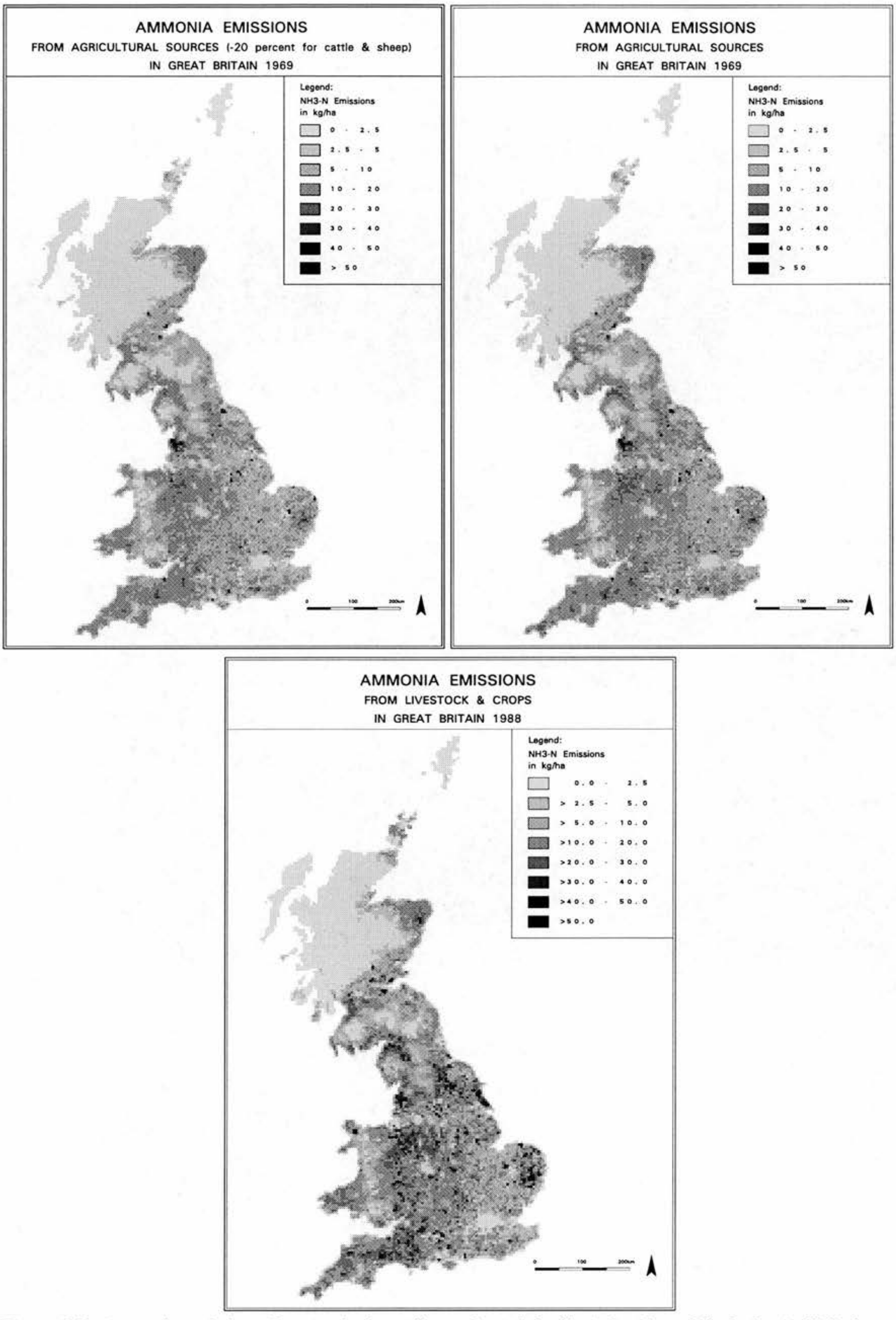


Figure 7.9. Ammonia emissions from agriculture (livestock and fertiliser) in Great Britain in a) 1969 (low emission scenario); b) 1969 (high emission scenario) and c) 1988; NB: emissions for 1969 and 1988 were mapped using the redistribution methodology of Hotson (1988) to ensure comparability).

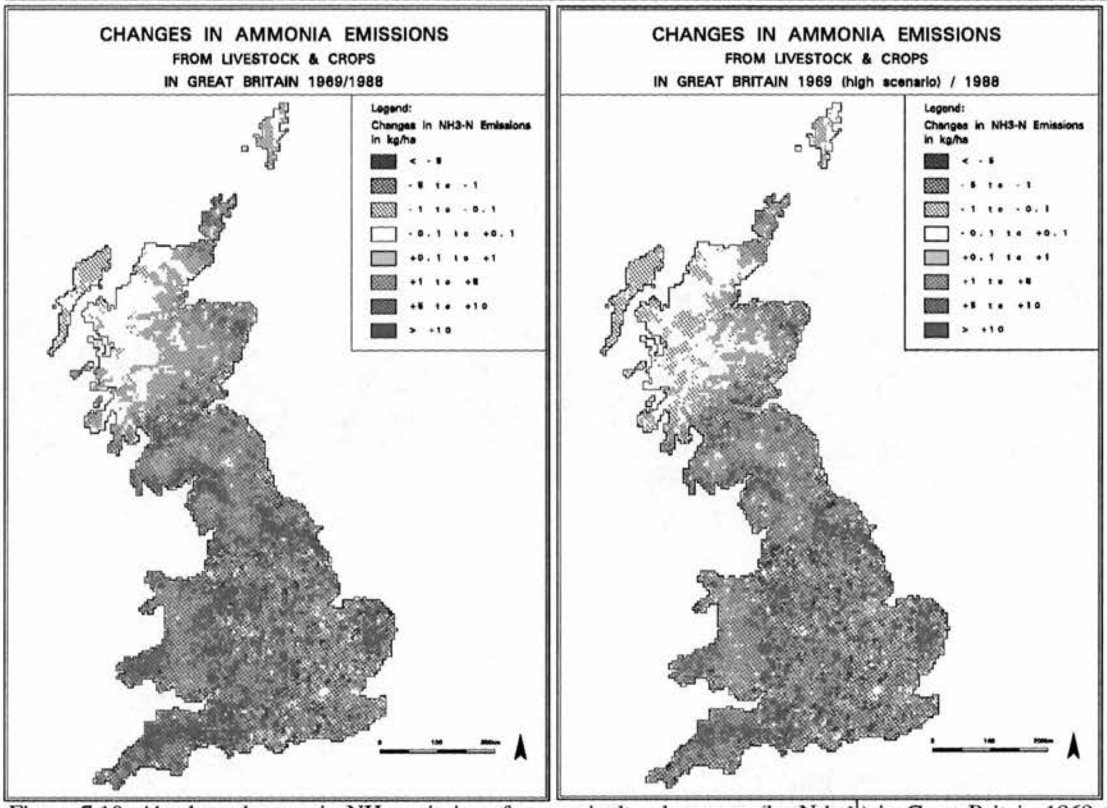


Figure 7.10. Absolute changes in NH_3 emissions from agricultural sources (kg N ha^{-1}) in Great Britain 1969 to 1988: a) low emission scenario for 1969 (source strength estimates for cattle & sheep reduced by 20%), b) high emission scenario for 1969 (source strength estimates as for 1988); redistribution based on Hotson (1988).

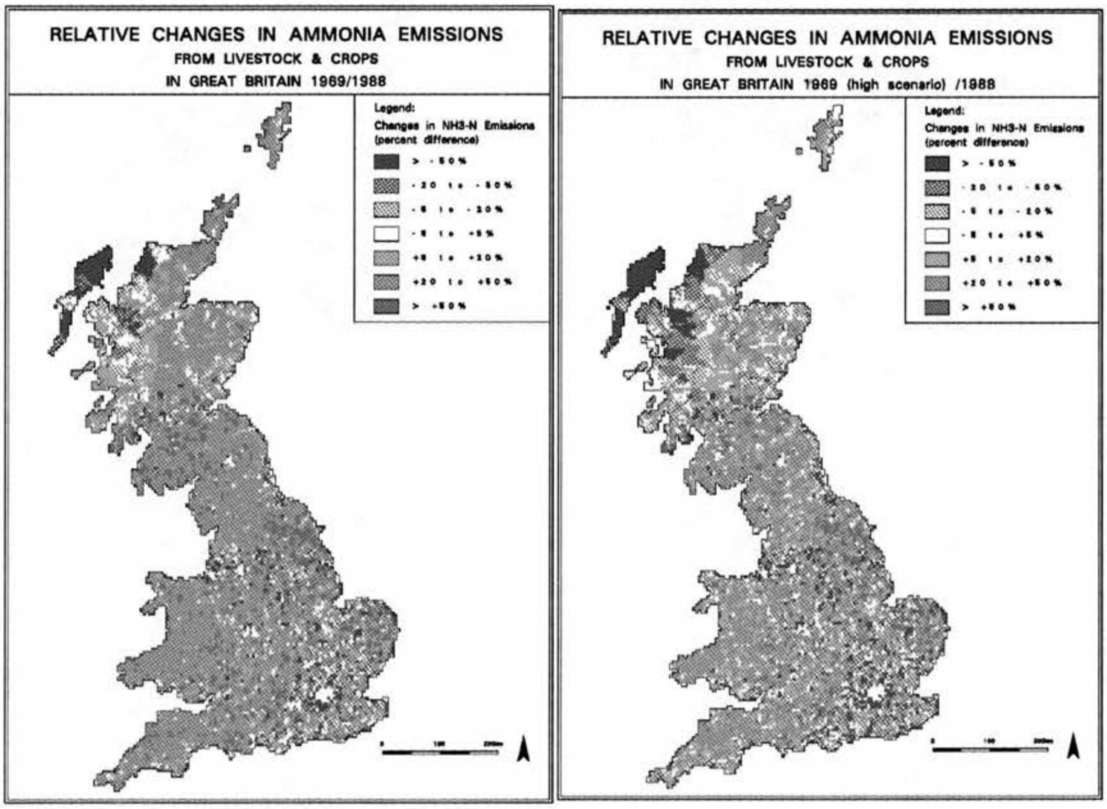


Figure 7.11. Relative changes in ammonia emissions from agricultural sources (in %) in Great Britain 1969 to 1988: a) low emission scenario for 1969 (source strength estimates for grazing livestock reduced by 20%), b) high emission scenario for 1969 (source strength estimates as for 1988); redistribution based on Hotson (1988).

It should be noted that the consistently large relative decreases in emissions for most of the Highlands and Islands of Scotland in Figure 7.11. are largely due to changes in the threshold size of farms required to participate in the Agricultural Census. As smaller farms were excluded from the census of all main holdings, the contributions of affected parishes to the total livestock numbers and crop areas declined. This change in the Census particularly affected north-western Scotland, where small farms are more frequently found than in other parts of Great Britain (SOAEFD, 1997; see Section 4.2.1.). Thus the decreased absolute and relative emissions estimated in north-west Scotland are to a large part caused by the underlying changes in the Census, rather than due to real changes in emission levels. (NB: This uncertainty could be eliminated, if spatially distributed data from the census of minor holdings for 1988 were made available for inclusion in the inventory).

The spatial trend in the magnitude of agricultural emissions between 1969 and 1988 is shown in Tables 7.6.-7.7. and Figures 7.12.-7.13. Emissions are estimated to have increased by more than 0.1 kg ha^{-1} in 63% and 74% of all GB gridsquares, respectively, using the high and low emission scenario for 1969 (Table 7.6., Figure 7.12.). Decreases by over 0.1 kg ha^{-1} have been estimated for 23% and 13% of all gridsquares, respectively, for the high and the low emission scenario. The most frequently occurring category in the analyses of both the high and the low scenario is in the range of 1-5 kg ha^{-1} (see Table 7.6. and Figure 7.12.).

Table 7.6. Absolute differences between 1969 and 1988 (in kg ha^{-1}) for spatially distributed inventories of agricultural emissions in Great Britain; positive values indicate increased emissions from 1969 to 1988, negative values indicate decreased emissions.

	High emission scenario for 1969		Low emission scenario for 1969	
Category (kg N ha^{-1})	Area (km^2)	Difference	Area (km^2)	Difference
>+50	400	0.2%	425	0.2%
+30 to + 50	900	0.4%	1,125	0.4%
+20 to +30	1,700	0.7%	2,075	0.8%
+10 to +20	9,825	3.8%	15,850	6.2%
+5 to +10	27,600	10.8%	42,225	16.5%
+1 to +5	78,825	30.8%	84,525	33.1%
+0.1 to +1	40,725	15.9%	42,850	16.8%
+0.1 to -0.1	36,950	14.4%	33,625	13.1%
-0.1 to -1	32,225	12.6%	18,025	7.0%
-1 to -5	23,675	9.3%	13,400	5.2%
-5 to -10	2,275	0.9%	1,100	0.4%
-10 to -20	450	0.2%	325	0.1%
-20 to -30	100	0.04%	100	0.04%
-30 to -50	50	0.02%	50	0.02%
< -50	25	0.01%	25	0.01%

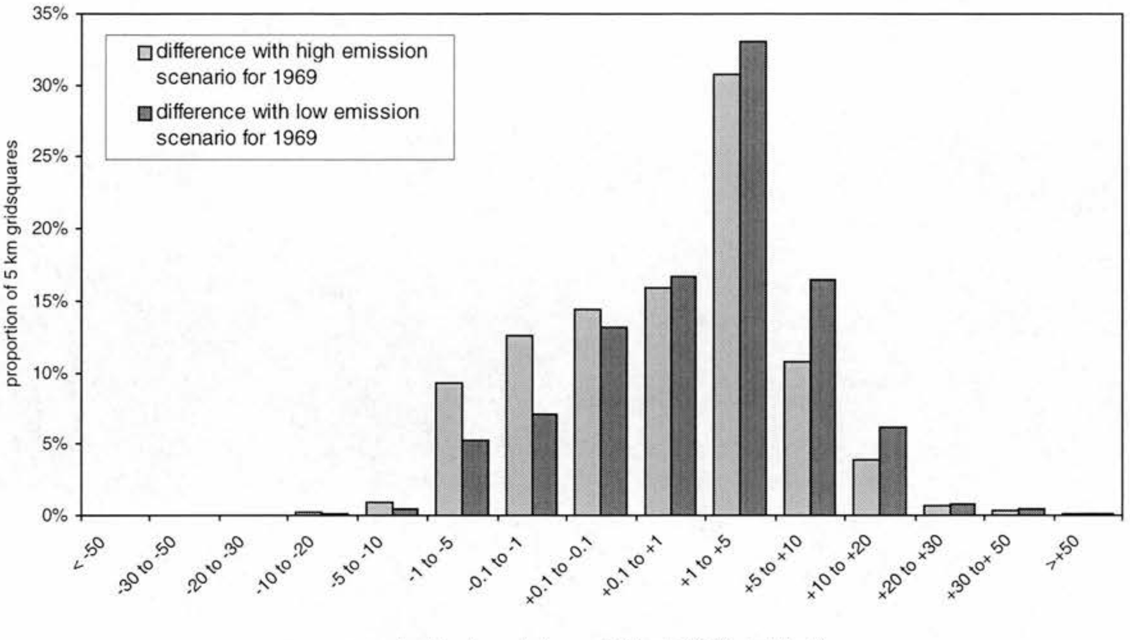


Figure 7.12. Absolute changes in ammonia emissions (in kg ha^{-1}) between 1969 and 1988 in Great Britain, analysed for both the low and the high emission scenarios for 1969, at a 5 km grid resolution. Positive values indicate increased emissions from 1969 to 1988, negative values indicate decreased emissions.

Relative changes in emissions between the two base years are shown in Table 7.7. and Figure 7.13., using both the high and the low emission scenario for 1969. For areas with increased emissions the magnitude of the changes is much more pronounced for the 1969 low emission scenario than for the 1969 high emission scenario. For areas with decreased emissions in north-western Scotland, the relative difference between the 2 years is larger with the high emission scenario. This is because the contribution of minor holdings in 1969 takes a bigger influence in the comparison with higher source strength estimates.

Table 7.7. Relative differences between 1969 and 1988 (in %) for spatially distributed inventories of agricultural emissions in Great Britain; positive values indicate increased emissions from 1969 to 1988, negative values indicate decreased emissions.

	High emission scenario for 1969		Low emission scenario for 1969	
Category (% difference)	Area (km^2)	Difference	Area (km^2)	Difference
< -50%	18,850	7%	13,075	5%
-20 to -50%	22,650	9%	10,425	4%
-5 to -20%	24,725	10%	14,075	6%
-5 to +5%	29,525	12%	21,550	9%
+5 to +20%	57,825	23%	38,000	15%
+20 to +50%	90,150	35%	133,075	53%
>+50%	12,000	5%	23,250	9%

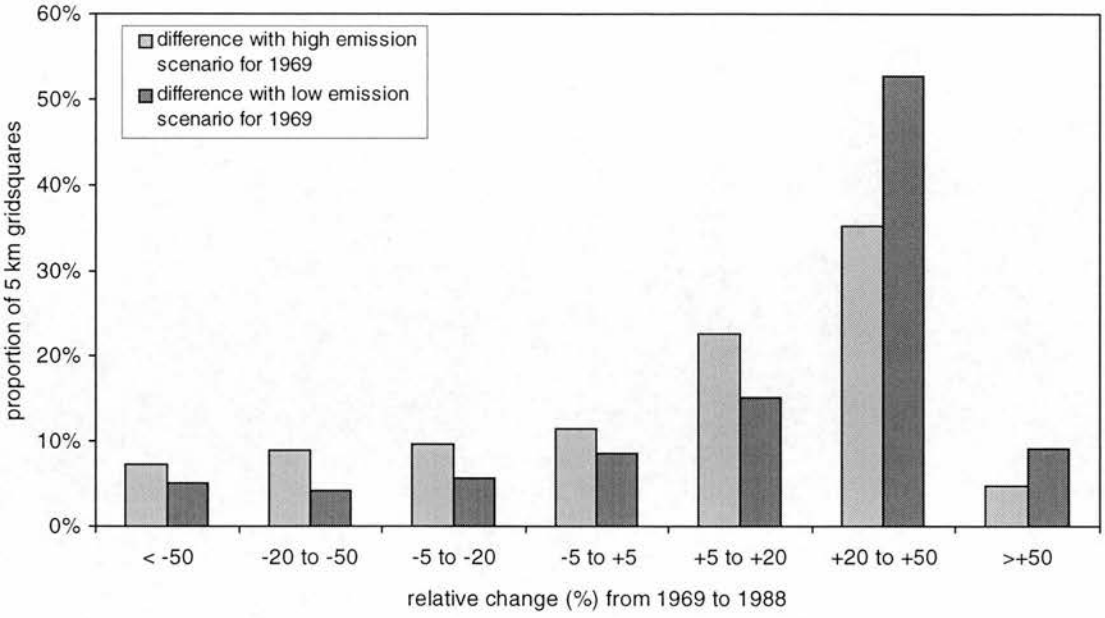


Figure 7.13. Relative changes in ammonia emissions (in %) between 1969 and 1988 in Great Britain, analysed for both the low and the high emission scenarios for 1969, at a 5 km grid resolution. Positive values indicate increased emissions from 1969 to 1988, negative values indicate decreased emissions.

7.4. ABATEMENT SCENARIOS

Ongoing negotiations in the UNECE Long-Range Transboundary Air Pollution Convention (LRTAP) use overall reductions within member countries to define goals for abatement. For the UK, for example, a maximum feasible scenario for abatement of NH_3 emissions was estimated to reduce emissions by 34%. A cost-optimised abatement scenario balancing several issues was estimated to result in a reduction of 20% (IIASA, pers. comm., 1997).

In reality, however, the benefits of abatement are not spread equally over the country, but vary depending on the spatial distribution of the abated sources. It is thus important to provide decision makers with a tool to model proposed abatement measures in spatially distributed scenarios. Following the scenarios through to deposition and effects estimates provides an insight into the expected improvements, regarding local and regional effects as well as long-range transport.

Ammonia emissions from livestock and crops vary between farms, depending on agricultural practice, N fertiliser input to arable crops, conserved and grazed grassland, as was discussed in detail in Chapters 2 and 3. The emission source strength estimates and other parameters used as model input data, which were described in Chapters 4 and 5, are assumed to be valid on average for the UK. Any significant changes in e.g. the type or amount of fertilisers used, the technique or timing of fertiliser application would result in changes in NH_3 emissions from crops and grassland. Similar assumptions can be made for emissions linked to livestock farming, such as landspreading techniques for manures, housing and manure storage.

Through comparison of emissions from different practices, the options leading to the smallest NH_3 emissions can be identified. In developing abatement strategies derived from these comparisons, however, the feasibility of these strategies has to be considered. Cowell and ApSimon (1998) have developed the MARACCAS model (Model for the Assessment of Regional Ammonia Cost Curves for Abatement Strategies) to assess the potential for abatement measures and to quantify the costs involved in implementing them. This is of considerable importance, as potential abatement measures have to be cost-effective in order to be successful.

If the decreases in emissions, predicted according to the current scenarios, are spatially redistributed, the expected benefits of different abatement measures can be modelled and mapped. Abatement scenarios were modelled in this study using proposed figures under implementation of Integrated Pollution Prevention and Control (IPPC) in the UK (EC, 1996). Decreases in NH_3 emissions of 4% for pigs and 7% for poultry are anticipated, if the suite of measures selected by MAFF in 1997 were implemented (Figure 7.14.). The measures chosen for this scenario do not achieve a large decrease in overall NH_3 emissions, although most of them can be applied at a relatively low cost. Implementing these measures would result in an emissions reduction of 2.9 kt N for the United Kingdom or 2.6 kt N for Great Britain. As the proposed abatement measures are aimed at the pig and poultry production sectors, which have the highest emissions on a per-area basis, the worst affected areas would be targeted. The effect of this would be a flattening of the very highest peaks of emission on the map for 1996. This would reduce the emission in the highest 1 km square by approx. 160 kg ha^{-1} (equivalent to a total reduction of 16 t in this square) in the model, although it should be noted that the 1 km estimates contain a very large uncertainty. In the highest 5 km square a reduction of 7 kg ha^{-1} is estimated (equivalent to a total reduction of 17.5 t in this square).

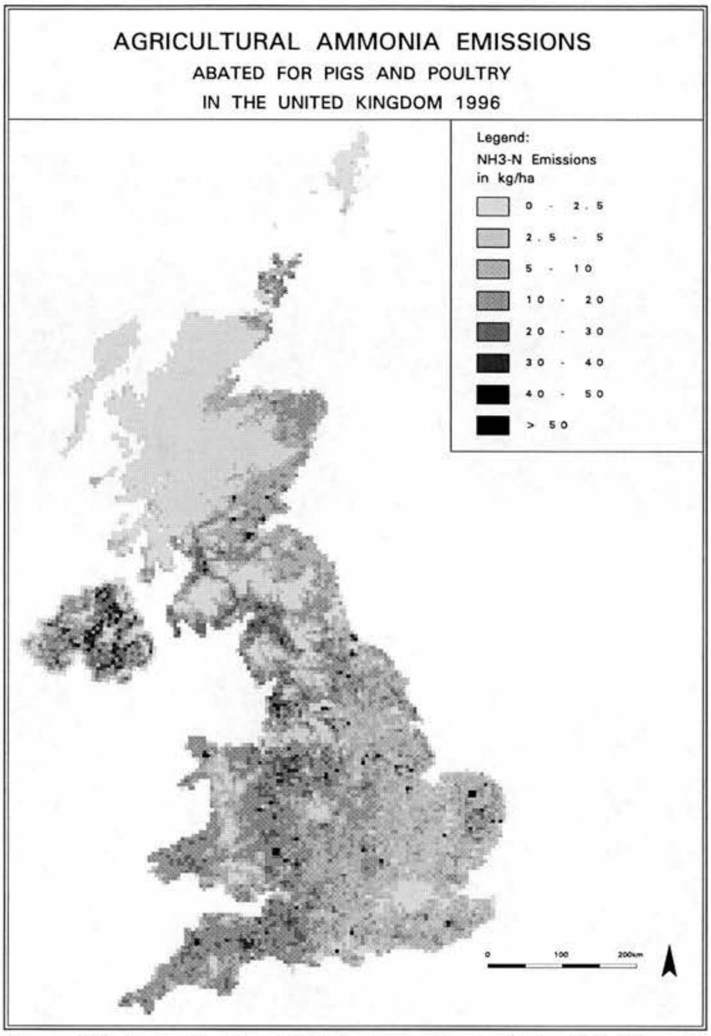


Figure 7.14. Scenario: abated NH_3 emissions in the United Kingdom in 1996 (-7% poultry, -4% pigs).

7.5. DISCUSSION AND CONCLUSIONS

This chapter has focused on changes in the magnitude and spatial pattern of British NH_3 emissions over time (1969, 1988, 1996), as well as made predictions of future changes regarding the implementation of abatement measures.

Temporal changes in NH_3 emissions from agriculture were analysed separately for two periods: 1969-1988 and 1988-1996. Increasing intensification and higher fertiliser N input per unit area of crops and grassland are estimated to have increased the total NH_3 emissions significantly from 1969-1988, as well as the spatial variability. The larger range of emission estimates in 1988 may be linked to the extensification of some marginal land on the one hand, and farm amalgamations as well as intensification on the other hand. The latter is especially relevant for intensive pig and poultry farming. Regarding the spatial pattern, the increase in the number of emission 'hot spots' is especially noticeable over this period, indicating an increase in the number of large intensive pig and poultry farms.

The total livestock numbers have not changed significantly during this period, except for sheep. Total sheep numbers nearly doubled between 1969 and 1988, mainly due to new market opportunities for British lamb in Europe and a higher lambing rate. Changes in livestock husbandry practice such as the increasing amount of fertiliser N applied to pastures and forage crops are well documented. This is estimated to result in higher emissions per animal in 1988 than in 1969, and thus in total emissions for the UK. More work is needed, however, to resolve the uncertainties in NH_3 source strength in the past in a quantitative manner.

During the period of 1988-1996, total estimated emissions from agricultural livestock in the UK have not changed significantly, while emissions from fertiliser use show a slight decrease. Changes in the spatial pattern of agricultural NH_3 emissions are more evident, with some significant decreases and increases for intensive livestock farming. These largest absolute changes are assumed to be due to relocation, new developments etc. of large pig and poultry enterprises, which provide the most intensive emission sources. Significant relative changes, both positive and negative, have also occurred in low emission areas, which represent relatively small absolute changes, compared with high emission areas. These changes may nevertheless be of importance regarding deposition and impacts for nitrogen-poor ecosystems in the vicinity of the sources. Summarising, it can be stated that, while only small absolute changes have occurred over most of the UK, considerable change has taken place in some areas. The latter can largely be linked to intensive pig and poultry farming.

The new model was also applied to simulate the introduction of abatement measures, and to predict the resulting changes in the spatial pattern of NH_3 emissions. The use of the model to provide scenarios of this type could be of significant benefit to decision makers. So far the work by IPPC was concentrated on pigs and poultry, as the focus was on large intensive installations from an industrial perspective. Sheep and cattle enterprises were not included under IPPC, although this was purely a political decision. Hence although the decisions were not based on environmental criteria, it turned out that IPPC is focusing on the appropriate source sectors. By concentrating on pigs and poultry, which provide the highest peaks in emissions, the benefits of abatement are focused on the most polluted areas.

Chapter 8
A local ammonia emissions inventory using field level data

8.1. BACKGROUND

8.1.1. Introduction

There are two main purposes for the development of a local scale inventory. Firstly, the spatial variability of NH_3 emissions can be investigated at a scale much finer than the 5 km grid resolution of the national inventory discussed in previous chapters (Chapters 4-7), in order to get an insight into the within-gridsquare variability. Secondly, a large scale local inventory with detailed information on agricultural practice enables comparisons between the national and local scales, regarding variability and uncertainty of emission source distribution and source strength estimates.

To some extent local inventories may be used to validate average conditions assumed for the national inventory, by providing an insight into the agricultural practice for an area at farm and field level. This may be compared with the average agricultural practice assumed for the national inventory. Furthermore, the availability of large scale/local data allows an assessment as to whether the redistribution process of parish census data and subsequent re-amalgamation to the 5 km grid level represents what is actually happening on the ground.

In the following sections, the study area and available data for the local (field-scale) inventory are described briefly, the methodology and results discussed, and variability and uncertainty aspects analysed.

8.1.2. The study area

The study area was selected, because of the availability of data on field level agricultural practice and because the land use within it represents the range of agricultural activities typical of lowland England. The area provides a wide range of emission sources, from livestock farming (cattle, sheep, poultry) to arable cropping and horticulture; and detailed information on agricultural practice was available from MAFF/ADAS (E. Lord, ADAS Wolverhampton, pers. comm., 1996). Information about the identity or location of the study area cannot be released due to the disclosivity issues involved, and some changes in field boundaries and livestock numbers have been made for this purpose where necessary. However, the aggregated results have been calculated from the original data and are not affected by this.

The total study area comprises over 3000 ha (approx. 5.5 km by 5.5 km), of which about 1600 ha is agricultural land. The non-agricultural land consists of mainly forest (conifer plantation and natural woodland or scrub) with some urban land. Approximately three quarters of the agricultural land are used for arable cropping, mainly in typical sandland rotations of barley or wheat with root crops (sugar beet and potatoes). The remainder is grassland, which is used for dairy, beef and sheep farming. The grass grazed by dairy cattle is stocked intensively and receives large inputs of N fertiliser. Within the area there is a large poultry unit (laying hens). The manure from this unit had formerly been disposed of within the study area, but following implementation of a local agricultural management scheme has been exported to well outside the area.

The local agricultural management scheme stipulated a maximum annual application of manure (175 kg ha^{-1} total N), and rules on timing for manures which release their N quickly. Most cattle and sheep farms had sufficient land for disposal of manure from their grazing animals at the introduction of the scheme, and the total quantity of manures applied was little changed. However, many pig and poultry farmers found that they had insufficient land for 'disposal' of manures within the scheme, and for the study area here the result was that all the poultry manure is exported from the area under the local agricultural management scheme.

8.1.3. Data sources

Detailed spatially distributed data were available for the study at the field level for 1993. For each agricultural field, annual fertiliser N application rates and, if applicable, livestock manure type and application rate were recorded. For fields under arable or horticultural cropping, information on the crops grown was available, and for grassland plots, detailed grazing and cutting records relating to the type of livestock grazed and the grazing months for each livestock type. The locations of the farmsteads with livestock housing and manure storage facilities were provided together with estimated numbers and categories of housed livestock.

A second dataset was made available for 1996 (E. Lord, ADAS Wolverhampton, pers. comm., 1998), in order to facilitate the comparison between the national inventory (1996) and local inventory. These data were, however, not sufficiently complete to enable the construction of a spatially distributed model for 1996. This is due to large errors in the spatial referencing of the data, which could not be resolved for the completion of this study. For this reason, the data records relating to the different fields had to be modelled in a tabulated (spreadsheet) format, thus not achieving the full benefits of a spatially distributed approach. For this study, however, this should not pose too large a problem, as the spatial variation within a sample 5 km grid square can be analysed equally well with the 1993 data, to provide data for local atmospheric transport and deposition models. The results for 1996 therefore had to be aggregated to the 5 km grid level to enable the comparison with the national inventory at a 5 km resolution.

8.2. METHODOLOGY

8.2.1. General issues

The model for 1993 was implemented in a Geographical Information System (ArcView GIS), which facilitates spatially distributed modelling by integrating the geographical references with the descriptive data for each area (field) of the map. Tabular access to the attribute data for each field in the study area is provided efficiently. Due to the availability of very detailed attribute data for each spatial unit (field), it was possible to be very specific about the model parameters and rules employed to calculate total emissions. For instance, it was possible to take specific fertiliser N application rates into account for modelling grazing emissions, dependent on stocking densities for each separate field. Emissions were calculated separately for the following component emission inventories:

- emissions from the application of mineral fertilisers to crops and fertilised cut grassland

- emissions from livestock grazing
- emissions from the landspreading of livestock wastes
- emissions from livestock housing and waste storage

These component inventories were subsequently incorporated into a total NH_3 emissions inventory for agricultural sources. In the following sections, the assumptions and modelling equations for the different parts of the inventory are explained.

Livestock numbers for both years were only suitable for the calculation of emissions from livestock housing and manure storage. They are not valid for the modelling of grazing and manure spreading emissions, as some of the farms have considerable grazing land outside the study area. Thus, not all the animals housed in the study area graze within its boundaries, and their manure may be spread outside the study area as well as inside. Other farms, in contrast, have their livestock housing units situated outside the study area, but their animals may graze on fields within, or their manure may be applied to fields within the study area. This issue was resolved by using the information provided for each field under grass, i.e. fertiliser input, grazing duration etc. (see Section 8.2.3.).

Compared with the 1993 data, the 1996 agricultural livestock data were recorded in a slightly different format. Livestock housing data are much more detailed for 1996, i.e. the numbers for the different categories of livestock and the housing duration were no longer based on estimates, but surveyed in detail at each farm. In contrast, the data recorded for each field of grassland for 1996 no longer contain explicit grazing records detailing the animal type and grazing cycles, but only the number of times the sward was cut for silage or hay making. Thus, a number of assumptions have to be made to derive grazing emissions from these data (see Section 8.2.3.).

These changes in data quality as well as data format make a comparison of the results between tabulated results for 1993 and 1996 very difficult and uncertain, and impracticable on a field by field basis. Thus this was not attempted, as the important comparison is between the 1996 field level inventory and the 1996 national 5 km grid inventory. The spatially distributed data for 1993 were treated in isolation, to show the spatial pattern of variability within a 5 km square regarding local source and sink areas. Nevertheless the data from both years, especially from 1996, also provide the opportunity for an analysis of the range of agricultural practices on the farms and fields in the study area, and for a comparison with the 'average' conditions assumed in the national inventory for 1996.

8.2.2. Emissions from the application of mineral fertilisers to crops and conserved grassland

NH_3 emissions were calculated for all fields where crops or grass for cutting were grown, using the same methodology for 1993 and 1996. It was assumed that fertiliser emissions from grazed swards were included in the grazing emissions (see 8.2.3.). This would not be true for emissions from the application of urea, which would be substantially higher. Therefore the emissions are most likely underestimated in the present model for fields receiving urea N. There was, however, no information available on the type of fertiliser used or the use of urea.

Emissions from fertilised fields were assumed to be proportional to the amount of mineral N fertiliser applied (Equation 1). As no information on the type of N fertiliser (ammonium nitrate, urea etc.) or method of application was available, a single volatilisation factor of 2.94% of the applied N was used, derived from the official NH_3 emission figures of DoE (1995). This is the same emission rate as in the national inventory.

$$E_{\text{fert}} = F_{\text{N}} * 0.0294 \qquad [1]$$

where E_{fert} is the emission from fertiliser application to crops and non-grazed grassland (in kg NH_3-N ha^{-1} $year^{-1}$) and F_{N} is the fertiliser application rate (in kg N ha^{-1} $year^{-1}$).

8.2.3. Emissions from livestock grazing

(a) 1993

It was necessary to develop different methodologies for the estimation of grazing emissions for 1993 (a) and 1996 (b), due to the different formats in which the data were provided. Emissions from grazing livestock for 1993 were calculated using the grazing records for different fields in the dataset available for this study. These records provided information on the livestock type and the number of grazing cycles the animals spent on the field.

Since most farms are not entirely within the study area (see Section 8.2.1.), the numbers of grazing animals for each field had to be estimated dependent on the size of each field and the fertiliser N application rate. It was necessary to make several assumptions regarding the stocking densities and the length of the different grazing periods for the fields:

- The grazing records indicate the succession of different livestock types (cattle, sheep) on the sward over the grazing period. It was assumed that each grazing cycle lasted for approximately 1 month (E. Lord, ADAS Wolverhampton, pers. comm., 1996). From this information grazing days for each field and livestock type were derived.
- Ammonia emissions and stocking densities on each field are calculated on the basis of the amount of fertiliser applied to the grassland (see below).

(i) Sheep

It was necessary to proceed through two steps to estimate sheep grazing emissions:

1) The stocking density (number of sheep ha^{-1}, S_{sheep}) was derived from experimental data at IGER (Jarvis *et al.* 1991, Jarvis and Pain 1990), dependent on the fertiliser application rate to each field (F_N):

$$S_{sheep} = 13 + 0.01786 * F_N \qquad [2]$$

2) The emissions per animal (e_{sheep} in g animal^{-1} day^{-1}) were derived from the same source:

$$e_{sheep} = 0.5 + 0.00167 * F_N \qquad [3]$$

The annual emissions from sheep grazing for each field (E_{sheep} in kg NH_3-N ha^{-1} year^{-1}) were estimated as follows:

$$E_{sheep} = e_{sheep} * S_{sheep} * d / 1000 \qquad [4]$$

where d is the number of days when sheep were assumed to be grazing that field, derived from the grazing records. This approach assumes that no NH_3 is emitted from a field following grazing, i.e. the emissions stop once the animals are removed from the field. Since some further emissions would be expected, emissions from livestock grazing can be assumed to be underestimates.

(ii) Cattle

For grazing cattle, Jarvis and Pain (1990) established the following relationship for average annual loss (NH_3-N ha^{-1} year^{-1}), which takes into account increasing stocking densities and increasing loss with increasing fertiliser N application (Equation 5):

$$E_{\mathrm{cattle\_avg}} = 0.000347 * F_{\mathrm{N}}^{1.854} \qquad [5]$$

No distinction is made between beef and dairy cattle. An implicit assumption that cattle are outside for 180 days per year in Equation 5 allowed the following equations to be derived: Equation 6 provides daily grazing emissions per animal (e_{cattle}, in kg ha^{-1} day^{-1}) during the grazing season. Annual emissions per animal (E_{cattle}, in kg NH_3-N ha^{-1} year^{-1}) are based on the time (in days, *d*) the animals were grazing on any particular field (Equation 7):

$$e_{\mathrm{cattle}} = 1.9278 * 10^{-6} * F_{\mathrm{N}}^{1.854} \qquad [6]$$

$$E_{\mathrm{cattle}} = e_{\mathrm{cattle}} * \mathrm{d} \qquad [7]$$

(b) 1996

Emissions from grazing livestock for 1996 were calculated in a slightly different way from 1993. As data on the type of livestock and the number of grazing periods were no longer available, the relevant information had to be derived from other sources. More assumptions were necessary in addition to the ones made for 1993, to enable the estimation of NH_3 emissions from grazing animals for 1996:

- The livestock types present on each farm determine the livestock types that may graze the fields belonging to each farm within the study area. The decision which animals were grazed on which fields had to be based on the most likely scenario of average farming practice. For fields with fertiliser N applications of >250 kg ha^{-1} dairy cows were assumed as the most likely grazing animals, and other cattle (beef cattle, dairy followers) and sheep were assumed to share the rest of the pastures (E. Lord, ADAS Wolverhampton, pers. comm., 1998). It is common practice in the area for sheep and cattle to be grazing on the same fields together or in rotation. Half the grazing period on these less highly fertilised fields was therefore assigned to sheep, and the other half to beef cattle and dairy followers.

- The length of the grazing period had to be derived from the number of times the fields under grass were cut for hay or silage making. In general, cuts are taken early in the season, when the grass growth is rapid and not all fields are needed to provide enough grazing for the farms' livestock (see Section 2.3.1.). After one or 2 cuts, the fields are normally used for livestock grazing for the rest of the season. The grass removed per cut represents approximately 40, 70, 90 and 100% of the total seasonal growth for 1, 2, 3 and 4 cuts respectively (E. Lord, pers. comm., 1998). Due to the seasonal variation in grass growth (see Section 2.4.2., Figure 2.9.), a hectare-month of grazing in August will provide sufficient forage for fewer animals, and consequently cause smaller NH_3 emissions, than a hectare-month in April. Thus, one cut leaves about 60% of the total grass on a field for grazing, which is equivalent to 4 months grass growth in a 6 month season. Two cuts, the maximum occurring in the study area on farms with grazing livestock, were assumed to leave enough grass growth to provide about 2 months grazing.

With the assumptions above providing the basis for the estimation of grazing periods for each livestock type, the same equations as for 1993 could be used for 1996. This applies to both sheep and cattle emission estimates.

8.2.4. Emissions from the landspreading of livestock manures

A major source of NH_3 emissions in the study area is the landspreading of livestock wastes. The datasets for both years provided detailed information regarding the type of manure applied to each field, the estimated N content and the application rates. Table 8.1. summarises the types of farmyard manure and application rates used in the study area for both 1993 and 1996.

Table 8.1. Manure types and application rates to fields in the study area for 1993 and 1996, summarised for manure types applied and crop types grown according to the field level data provided for the two years; the level of application rates is classified in relation to the N content of the different manure types and the application rate.

Manure type 1993	Dry Matter (%)	Crop	Application rate (t ha^{-1})	Level of application rates
Cattle	25%	Grass	12	Low
Cattle	25%	Sugarbeet	16-37	Medium-High
Broiler	60%	Linseed	7	Medium-High
Manure type 1996	**Dry Matter (%)**	**Crop**	**Application rate (t ha^{-1})**	**Level of application rates**
Cattle	25%	Grass	25	Medium
Cattle	25%	Potatoes	10	Low
Cattle	25%	Flowers	10	Low
Cattle	25%	Sugarbeet	20-35	Medium-High
Broiler	60%	Carrots	4	Medium

It is assumed that the manure is spread uniformly over the fields not rapidly incorporated into the soil. Different estimates have to be calculated for manures produced by different livestock types:

(a) Cattle

The only type of cattle manure spread to fields within the study area was straw-based farmyard manure (FYM). The following relationships for cattle FYM were used:

- The TAN (total available N) content of cattle FYM is estimated at 0.6 kg t^{-1} (BBSRC, 1997a, b).
- 65% of the TAN in cattle FYM applied to grassland and arable land are estimated to be emitted as NH_3-N (BBSRC, 1997a, b).

This results in an estimated emission of 0.39 kg N t^{-1} of cattle FYM applied to land:

$$L_{\text{cattle}} = A_{\text{cattle\_FYM}} * 0.39 \qquad [8]$$

where L_{cattle} is the emission from the landspreading of cattle FYM (kg N ha^{-1} $year^{-1}$) and $A_{\text{cattle\_FYM}}$ is the manure application rate (in t ha^{-1} $year^{-1}$).

(b) Poultry

It should be noted that the poultry manure spread in the study area is broiler manure (according to the dataset used for this study), not manure from the laying hens at the large poultry unit within the study area. The source of this broiler manure is outside the boundaries of the study area.

The BBSRC (1997a,b) studies calculate emissions from the landspreading of poultry manure as 35% of the AUN (= Ammoniacal N and Uric Acid N) applied to fields. Table 8.2. shows typical estimates for different types of poultry manure. Broiler manure has a higher AUN content, as it is generally drier than layer manure.

Table 8.2. AUN content and emission estimates for landspreading of poultry manure (BBSRC, 1997a, b).

Manure type	Emission (% of AUN applied)	Average AUN in manure (kg t^{-1})	Emission (AUN in kg t^{-1})
Layers	35	7.5	2.625
Broilers	35	11.6	4.06

Applying the data in Table 8.2. to the information recorded for each field, the following equation for emissions from the landspreading of broiler litter (L_{broiler} in kg AUN t^{-1} ha^{-1}) can be implemented:

$$L_{\text{broiler}} = 4.06 * A_{\text{broiler}} \qquad [9]$$

where A_{broiler} is the manure application rate (in t ha^{-1}). For an application rate of 7 t of broiler waste per hectare, for instance (Table 8.1.), this results in estimated emissions of 28.4 kg ha^{-1} N.

8.2.5. Emissions from livestock housing and manure storage

On the four farms with livestock housing units in the study area, cattle and poultry are housed for at least part of the year. The assumptions and model parameters for NH_3 emissions from livestock housing and manure storage are explained below.

The livestock numbers for each farm are shown in Table 8.3. for 1993 and 1996. It should be noted that the animal numbers for 1993 are approximate and no information is provided on the housing period for cattle. Therefore, an average housing period of 6 months was assumed. All cattle (beef and dairy) are kept on straw-based FYM. Sheep are outdoors all year, therefore no emissions from sheep housing were included in the model.

Table 8.3a. Estimated livestock numbers and months of housing (in brackets) for farms in the study area in 1993 (E. Lord, ADAS Wolverhampton, pers. comm., 1996) NB: Livestock numbers were not provided as part of the local agricultural management scheme for the study area. These have been estimated independently.

Farm	Dairy cows	Other cattle	Sheep	Poultry-Layers
1	-	70 (6)	2,000 (0)	-
2	-	-	-	140,000 (12)
3	70 (6)	70 + 35 [a] (6)	2,000 (0)	-
4	110 (6)	100 + 55 [a] (6)	1,000 (0)	-

[a] On the dairy farms, additional dairy followers (estimated about half the number of the diary cows) were counted with the beef cattle for the purpose of calculating NH_3 emissions.

Table 8.3b. Livestock numbers for farms and months of housing (in brackets) in the study area in 1996 (E. Lord, ADAS Wolverhampton, pers. comm., 1998). NB. Some numbers have been changed to preserve confidentiality.

Farm	Dairy cows	Young dairy cattle	Young beef cattle	Sheep (adults)	Laying hens
1	-	-	90 (12)	340 (0)	-
2	-	-	-	-	450,000 (12)
3	850 (5)	180 (12)	-	1000 (0)	-
4	150 (6)	58 (0)	-	2100 (0)	-

Emissions from manure storage may be calculated using the BBSRC studies (1997a, b), dependent on the surface area of the storage facilities. However, data on manure storage were not available for the present study. Other authors calculate housing and storage emissions on a per-animal basis, which is more appropriate for the present study (e.g. TFEI, 1996; ECETOC, 1994; Asman, 1992b; Sutton *et al.*, 1995). The European Emissions Inventory Guidebook (TFEI, 1996) gives the estimates for the main livestock categories as shown in Table 8.4.:

Table 8.4. Housing and storage emission estimates for different livestock categories (TFEI, 1996; kg NH_3-N animal^{-1}).

Livestock categories	Housing (annual)	Storage (annual)	Housing & storage (annual)	Housing & storage (monthly)
Dairy cows	7.20	3.17	10.37	1.73
Other cattle (young cattle, beef cattle, suckler cows)	3.60	1.58	5.18	0.86
Laying hens	0.16	0.03	0.19	0.016

For dairy cows, the housing and storage emission estimates (TFEI, 1996) are based on the animals housed for 180 days during winter and for a couple of hours each day for milking during summer. However, this is not strictly true, as emissions are expected to continue from the milking area after the dairy cows are released back into the field. Therefore, emissions from the housing of dairy cows may be underestimated. The average storage duration for all cattle manures is assumed to be 6 months (TFEI, 1996).

For the 1996 dataset, exact housing durations for each livestock class on each farm were available. Therefore the housing emissions were adjusted for the real housing duration in the 1996 model. It was assumed that variations in housing emissions due to variations in housing duration would be mirrored by similar variations in manure storage emissions. Thus, storage emissions were calculated in proportion to the housing duration.

8.3. RESULTS AND DISCUSSION

Emissions in the study area are highly variable, depending on the source strength present in each field in the study area (Figure 8.1.). Emissions for 1993 range from just under 1 kg NH_3-N ha^{-1} $year^{-1}$ for a field with low N fertiliser application rates to approximately 1700 kg NH_3-N ha^{-1} $year^{-1}$ from the intensive poultry unit. Except for the farmsteads themselves, where the emissions from livestock housing and waste storage are much higher, emissions are generally estimated as below 30 kg NH_3-N ha^{-1} $year^{-1}$.

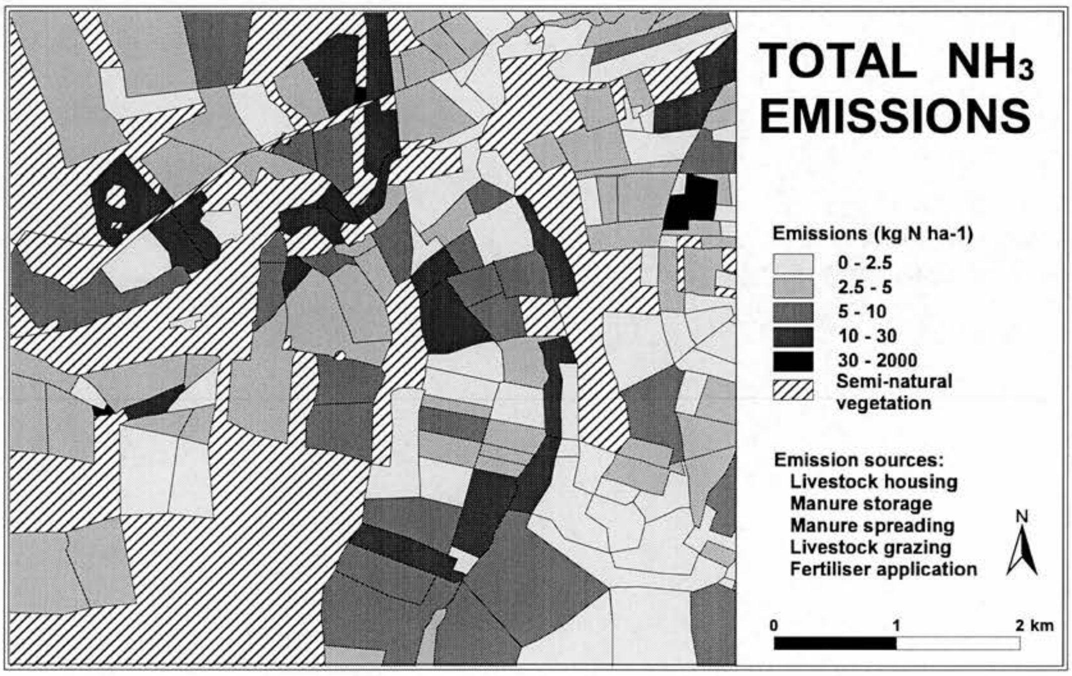


Figure 8.1. Total emissions from agricultural sources in the study area for 1993. NB. Some field boundaries and emission values have been changed to preserve confidentiality.

The magnitude and variability of emissions calculated for 1996 for each field are very similar to the results of 1993, although the values for each single field vary between the years due to crop rotation, and, for grazed fields, also due to the difference in the methodology applied. The only exception to this is a large increase in housing and storage emissions for 3 out of the 4 farmsteads (see Section 8.3.4.).

8.3.1. Emissions from the application of mineral fertilisers to crops and cut grassland

A large proportion of the agriculturally used area receives mineral N fertiliser applications. Emissions from the application of mineral N fertiliser to crops and non-grazed grassland are estimated to be under 11 kg NH_3-N ha^{-1} $year^{-1}$ in all cases for 1993 and 1996, with most fields estimated to emit less than 5 kg NH_3-N ha^{-1} $year^{-1}$ (Figure 8.2.). Emissions from the application of fertiliser to grazed grasslands are included with the grazing emissions.

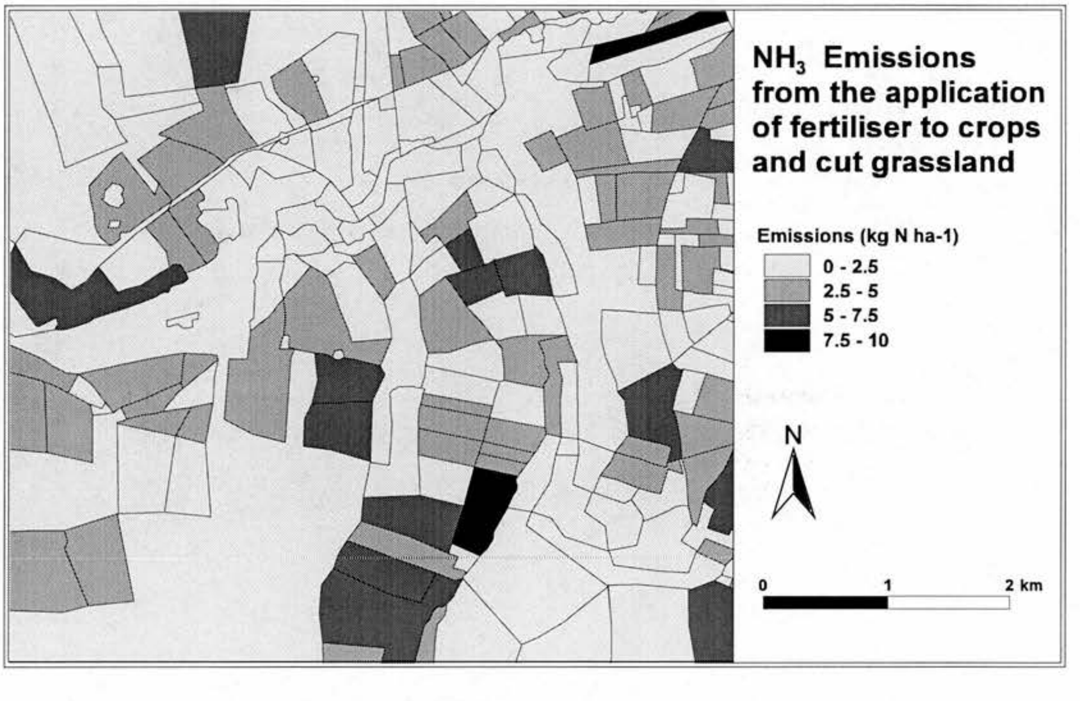


Figure 8.2. Emissions from the application of mineral N fertiliser to crops and non-grazed grassland in the study area for 1993. NB. Some field boundaries and emission values have been changed to preserve confidentiality.

8.3.2. Emissions from livestock grazing

Emissions from livestock grazing estimated for 1993 varied between about 1 kg NH_3-N ha^{-1} $year^{-1}$ and 3.6 kg NH_3-N ha^{-1} $year^{-1}$ for sheep and are shown in Figure 8.3. The highest value was estimated for sheep grazing a field with 336 kg N application for 6 months at an average stocking density of 19 sheep per hectare. In

1996, sheep emissions were much lower on each field grazed by sheep, due to the assumptions that had to be made because grazing records were no longer available. This resulted in sheep grazing only on fields with less than 250 kg ha^{-1} fertiliser N input, and sheep sharing their pastures with young cattle.

For cattle the maximum grazing emission in 1993 was estimated at 25 kg NH_3-N ha^{-1} $year^{-1}$ for a field with 420 kg N ha^{-1} applied and grazed intensively for 6 months. In 1996, the maximum grazing emissions per hectare grazed grassland were even higher: Several fields receiving 450 kg N ha^{-1} were estimated to be grazed intensively by dairy cows for 6 months, resulting in emissions of 28.8 kg N ha^{-1}. If animals were on a field for only a short time rather than during most of the grazing season, the total emission from this field is likely to be an underestimate, due to continuing volatilisation after the animals were moved from the field.

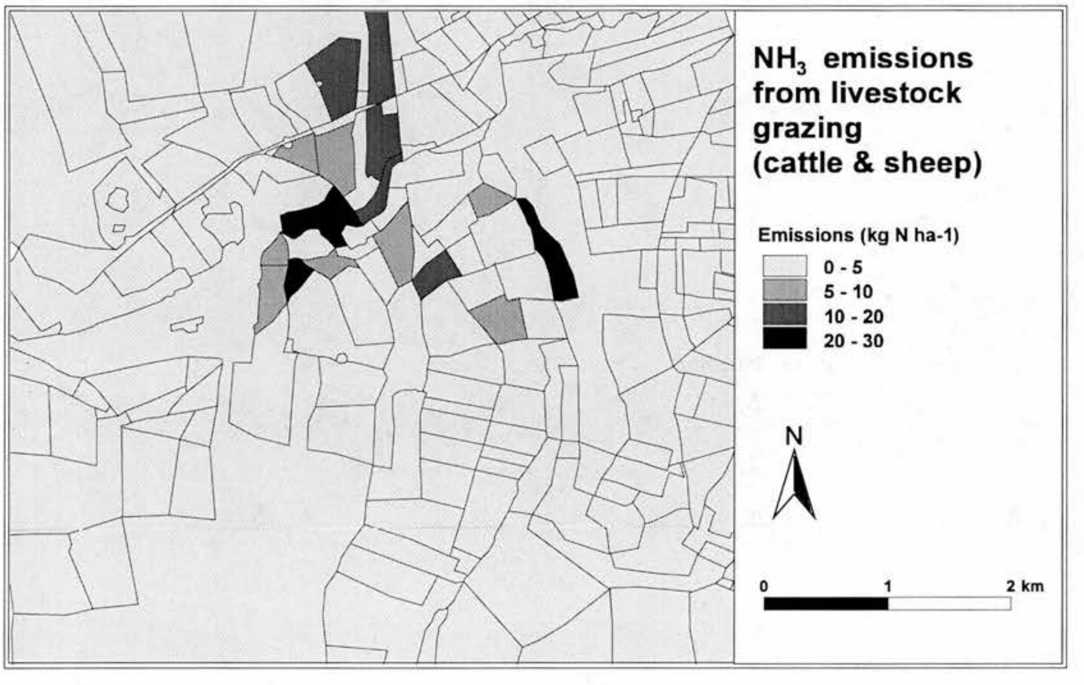


Figure 8.3. Emissions from livestock grazing in the study area for 1993. NB. Some field boundaries and emission values have been changed to preserve confidentiality.

8.3.3. Emissions from the landspreading of livestock manures

Figure 8.4. shows emissions from the landspreading of livestock manures estimated on the basis of manure type and application rates. The higher available N content in poultry waste resulted in larger emissions than from the less concentrated cattle FYM, which was spread at higher application rates. Maximum emissions are

estimated to occur at approx. 28 kg NH_3-N ha^{-1} $year^{-1}$ in 1993 and 16 kg NH_3-N ha^{-1} $year^{-1}$ in 1996.

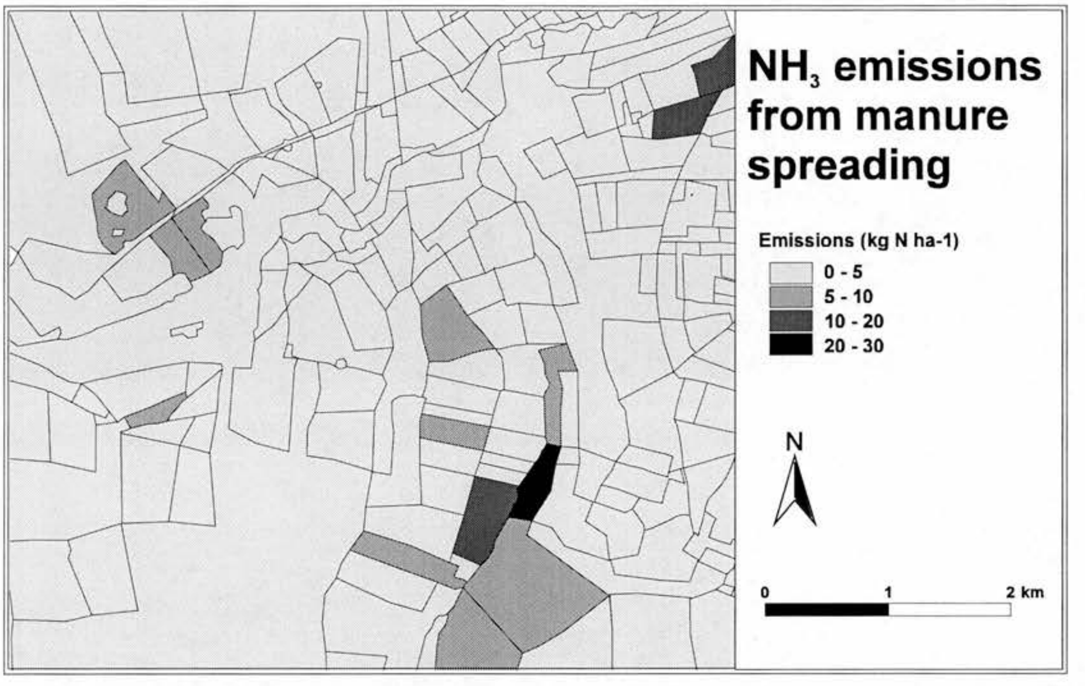


Figure 8.4. Emissions from the landspreading of livestock manures in the study area (in kg NH_3-N ha^{-1} $year^{-1}$) for 1993. NB. Some field boundaries and emission values have been changed to preserve confidentiality.

8.3.4. Emissions from livestock housing and manure storage

The emission totals for livestock housing and manure storage for the 4 farmsteads in the study area were calculated for 1993 and 1996, using the numbers in Tables 8.3. and 8.4. (above). The spatially distributed results for 1993 are not shown in a separate map, but included into Figure 8.1. Table 8.5. shows the results for each farm in kg NH_3-N $farm^{-1}$ $year^{-1}$ and kg NH_3-N ha^{-1} $year^{-1}$ for the area occupied by the farm buildings and stores (the farm area having been estimated approximately on the digital map provided).

The emissions from housing and storage appear to have changed dramatically for all 4 farms. For the poultry farm, this is due to an expansion of the housing capacity to 3 times the size of 1993. Large real changes may have occurred not only on the poultry farm, where the bird numbers are confirmed values for both years, but also on the cattle farms. However, it may be suggested that the changes in cattle housing emissions could be artefacts, due to the livestock data for 1993 being estimates rather than directly surveyed. This not only relates to total numbers and types of livestock, but also to the housing duration. For instance, the cattle on Farm 1 were housed all year in 1996, whereas an average housing period of 6 months was assumed for 1993. An analysis of the results for both years showed that an average housing duration of 6 months for all cattle in 1996 results in an overestimate for dairy cows (~ 15%), and in an underestimate for other cattle (~40%).

Table 8.5a. Housing and storage emissions (units: kg NH_3-N farm^{-1} year^{-1} unless specified otherwise) for the farms in the study area in 1993.

Farm	Dairy cows	Other cattle	Laying hens	All livestock	Farmstead area (ha)	Total emissions (kg N ha^{-1} year^{-1})
1	-	363	-	363	1.1	330
2	-	-	22,680[a]	22,680	13	1,745
3	726	544	-	1,270	1.2	1,058
4	1,141	803	-	1,944	1.2	1,620

[a]90% Occupancy of the poultry houses is assumed (E. Lord, pers. comm., 1996).

Table 8.5b. Housing and storage emissions (units: kg NH_3-N ha^{-1} farm^{-1} unless specified otherwise) for the farms in the study area in 1996.

Farm	Dairy cows	Other cattle	Laying hens	All livestock	Farmstead area (ha)	Total emissions (kg N ha^{-1} year^{-1})
1	-	932	-	932	1.1	847
2	-	-	76,950[a]	76,950	13	5,919
3	7,345	1,865	-	9,210	1.2	7,675
4	1,556	-	-	1,556	1.2	1,296

[a]90% Occupancy of the poultry houses is assumed (E. Lord, pers. comm.).

8.4. COMPARISON OF THE LOCAL INVENTORY WITH THE NATIONAL INVENTORY FOR 1996

8.4.1. Comparison of the local and national inventories for 1996

The study area provides a test case for investigating the spatial variability of NH_3 emissions within a sample 5 km gridsquare of the national inventory. It also presents an opportunity for comparing emission estimates for an area, which were computed with a field-specific methodology and greater levels of detail regarding source distribution and emission source strength. This case study was possible because data for the same base year (1996) were made available for models at both the national and local scale. Both methodologies, the national and the local model, are dependent on sufficiently detailed information regarding agricultural practice for the desired resolution of the respective inventories.

The dataset also provides an insight into the specific agricultural practice for an area at farm and field level. This allowed an assessment of the quality of the 'average' situation, which had to be assumed for the development of the national inventory.

Three scenarios were derived to facilitate the comparison between the local and national inventory, regarding the results as well as the model parameters regarding agricultural practice. For this purpose, the relevant 5 km gridsquare in the national inventory had to be identified and analysed. The size and shape of the study area make it ideal for a comparison with a 5 by 5 km grid square. However, it overlaps 4 different grid squares in the 5 km national inventory. It was therefore necessary to aggregate the relevant grid squares in the 1 km inventory to derive a 5 km sample square more or less congruent with the study area. The total amount of NH_3 emitted within the 5 km grid square congruent with the study area and the average emission rates in kg N ha^{-1} are given in Table 8.6. (Scenario 3).

Table 8.6. Emissions for the study area in 1996: comparison between the local inventory and the relevant 5 km by 5 km gridsquare of the national inventory. Scenario 1 was derived from the local scale inventory described in this chapter, by adding up the emissions estimated for each field for the whole study area. Scenario 2 was calculated by applying the average emission source strength data used in the national inventory to the livestock numbers and crop areas recorded in the local inventory. Scenario 3 represents the results of the national 5 km inventory for the gridsquare congruent with the local study area. The differences in the total area between the national 5 km gridsquare and the irregularly shaped local study area were accounted for by normalising the results on an area-basis.

	Local inventory (3000 ha area)		**Normalised results**	National inventory (2500 ha area)	**Normalised results**
Emission source	Scenario 1 (kg NH_3-N)	Scenario 2 (kg NH_3-N)	**Avg. emissions (kg NH_3-N ha^{-1} yr^{-1}) for the study area**	Scenario 3 (kg NH_3-N)	**Avg. emissions (kg NH_3-N ha^{-1} yr^{-1}) for 5 by 5 km cell**
Fertiliser	4,587	4,547	**1.5**	5,256	**2.1**
Livestock	92,629	92,531	**30.9**	80,262	**32.1**
Agriculture	97,216	97,078	**32.4**	85,518	**34.2**

In order to compare the emissions from the 5 km national inventory gridsquare (Scenario 3) with the local scale inventory, the total NH_3 emission estimates for each field in the study area were added up. Normalised emission estimates were derived by dividing the total emissions from the study area by the total area (Scenario 1 in Table 8.6.).

A third estimate (Scenario 2) was calculated by applying the same average emission source strength data as in the national inventory (Chapter 5) to livestock numbers for the 4 farms with livestock housing within the study area. The average fertiliser N application rates from the BSFP for England and Wales (Burnhill *et al.*, 1997) were applied to the fields in the study area, instead of the individually recorded rates, to calculate fertiliser emissions.

For both categories, fertiliser and livestock emissions, the average emissions per hectare are very similar between the local study and the national estimate (Table 8.6. above; Scenarios 1 and 3). This merits closer investigation, considering the different level of detail and spatial resolution in the input data, the different methodologies applied in both the national and local studies and some large deviations from average farming practice in the local inventory.

Comparing the two estimates for the local inventory first, the following can be observed: both totals for the livestock as well as fertiliser emissions agree closely. This is surprising, considering that Scenario 2 was calculated under very simplified assumptions, i.e. that only livestock registered with farms 1-4 contributes to emissions within the study area, with no other farms contributing to the total. It was also assumed that all manure from the poultry farm (layers) was spread within the area boundaries in Scenario 2.

A comparison of fertiliser emissions in the study area showed several differences between the averages for each crop type derived from the field scale data in the study area and the averaged BSFP estimates for England & Wales (Burnhill *et al.*, 1997). Potatoes and cut grassland received higher N applications by 75 and 80 kg N ha^{-1}, respectively, in the study area than the average values estimated for these crops in the BSFP tables. This difference between the averaged national and the local agricultural practice is, however, balanced by the lower N input to winter barley (-35 kg ha^{-1}) and winter wheat (-40 kg ha^{-1}) in the study area. Average fertiliser application rates to the other main crops within the study area, sugar beet and spring barley, are very close to the BSFP averages (Table 8.7.). The good match between the 2 estimates can therefore be attributed partly to the smoothing effect of aggregating all crop emissions together, and partly to the fact that most farmers were applying fertiliser to their crops at more or less the average rate given in the BSFP tables.

Another simplification applied in the national inventory is that roughly one third of the improved grassland is conserved as hay or silage. Within the study area, 45% of the improved grassland is cut, with 2 cuts on average in 1996. Two cuts are equivalent to 2/3 of the season's growth (67%), and so the total amount of grass dry matter removed by cutting is 45% * 67% = 30% of the total growth. Thus, the average conditions assumed for the UK in general (Scenarios 2, 3) appear to hold for the study area.

Table 8.7. Analysis of fertiliser application rates to crops and grassland, comparing data for the study area in 1996 and average fertiliser application rates (BSFP: Burnhill *et al.*, 1997) for England and Wales 1996.

Study area					BSFP 1996	
Crop type	Crop area (ha)	Number of fields	Avg. N rate (kg ha^{-1})	Range (kg ha^{-1})	Crop /crop group	Avg. N rate (kg ha^{-1})
Bulb onions	14	1	100	100	Other tillage	63
Carrots	75	7	73	43-125	Other vegetables	96
Flax	6	1	108	108	Other tillage	63
Flowers	9	1	100	100	Other tillage	63
Forage maize	2	1	0	0	Forage maize	52
Linseed	26	6	66	0-75	Linseed	53
Potatoes	132	16	247	238-269	Potatoes	174
Spring barley	125	17	97	88-125	spring barley	95
Spring wheat	26	3	12	72-140	wheat (spring & winter)	185
Sugar beet	260	35	106	0-135	sugar beet	107
Winter barley	319	41	103	0-135	winter barley	138
Winter wheat	100	12	148	95-188	wheat (spring & winter)	185
Grassland (total)	312	44	248	0-450	grassland **	-
Grassland (cut)	142 *	15	228	0-400	cut grass	149

* 2 cuts on average

** grazed grass included with livestock grazing emissions

The 5 km grid square covering the study area contains 3 whole parishes and parts of 8 others. The mean livestock NH_3 emissions of the local and national inventories are very similar at just over 30 kg N ha^{-1} (Table 8.6.), averaged over the whole area. This appears to confirm that the more general methodology applied for the national inventory and the data and assumptions used provide relatively stable results, at least for this part of the country.

For most of the livestock emission sub-sources (housing & manure storage, grazing and landspreading of manures), the emissions in the field level inventory were derived separately and independently from the source strength estimates used in the national inventory (Scenario 1). For instance, grazing emissions were calculated individually for each field, depending on the fertiliser N application rate. The N application rates to grazed grassland in the study area are very high, compared with average conditions according to the BSFP tables (Burnhill *et al.*, 1997), thus resulting in higher than average grazing emissions.

Housing and storage emissions in the study area (Scenario 1) were estimated to be higher than the national average (Scenarios 2, 3). For cattle, this was due to the majority of cattle housed in the study area being dairy cows (85%), which emit more for a given housing period than the 'average cattle' in the national inventory. A large proportion of the young cattle in the study area (85%) were housed all year round, which increases their total annual emissions significantly, compared with 'average cattle' that are housed for 6 months. Similarly, the housing and storage emissions for poultry were higher in the study area (Scenario 1) than the 'average poultry' of the national inventory (Scenarios 2, 3), since the birds concerned were laying hens (compare Section 3.2.4.). This together with the higher cattle housing and storage emissions offset the fact that the layer manure is exported from the study area in Scenario 1. This export of a large proportion of the emissions from the poultry farm in Scenario 1 is mainly responsible for bringing the total emissions from the study area down again.

However, if the study area had not been declared part of the local agricultural management scheme, the total livestock emissions in the local inventory (Scenario 1) would be significantly higher, without manure management regulations. Thus, it can be concluded that emissions from intensive agricultural areas, without regulations regarding manure management and with large intensive livestock enterprises, may be underestimated in the national inventory. This may be partially due to deviations from the average agricultural practice assumed in the national inventory (Scenarios 2, 3), such as a longer housing duration for intensive beef production. Underestimates in areas with intensive pig and poultry farming at the national level may also be due to the spatial distribution approach, which is limited by civil parishes as the basic data units. This results in the spatial distribution of housing, storage and landspreading over all potentially suitable land in the parish, rather than at the specific location of the farm, thus smoothing out emission 'hot spots' over the whole parish area.

From this comparison of different inventories for a high emission study area it can be concluded that the national inventory may overestimate for low emission areas and underestimate for high emission areas, due to the assumptions regarding average agricultural practice. In order to investigate this issue further, the acquisition of data with similarly detailed information for other areas of the UK is suggested. Suitable sample areas should include upland and hill farming areas, as well as other more intensive areas, e.g. a gridsquare with intensive pig farming. Local inventories, such as the one developed here, could then be used to assist in the derivation and calibration of a methodology to include spatially variable emission source strength estimates in the national inventory.

8.4.2. The spatial variability of emissions in the national and local inventories for 1996

In a second comparison, the spatial variability of NH_3 emissions in the national and local inventories is assessed. The averaged results per hectare hide a very high local variability of emissions within short distances. For instance, the emission estimated for fields with little or no fertiliser N application is close to zero, and is more likely to be a sink for atmospheric NH_3 than a source. On the other hand, each hectare of the poultry farm's housing and storage compound emits an estimated average of approx. 8,000 kg N per year in the local inventory. Thus, the national inventory at the 5 km scale on its own without additional information is probably smoothing out a large number of local 'hot spots' such as the poultry farm in the study area. Much larger intensive livestock enterprises in other areas of the country appear even more prominently on the 5 km map, despite being smoothed considerably by the aggregation to the 5 km level.

8.4.3. Further uncertainties in the model and results

It is likely that the present emission estimates of the local inventory, both for 1993 and 1996, are underestimating the total emissions for this area. This is expected due to two reasons:

- Emissions from non-agricultural sources were not included in this study. This does, however, not affect the comparisons undertaken above (Sections 8.4.1., 8.4.2.).
- Emissions from livestock grazing and housing are estimated only for the time animals are reported to be housed or grazing, which is a simplification. The fields from which the animals are removed after grazing probably continue to emit for some time, and the same is valid for livestock houses. Once the animals are put out onto the fields for grazing, emissions continue to occur within the housing area.

8.5. DEVELOPMENT OF AIR CONCENTRATION FIELDS, DEPOSITION AND CRITICAL LOADS MAPS FOR THE STUDY AREA

The impacts of the spatial variability of NH_3 emissions at the local level were investigated by predicting air concentrations and depositions with atmospheric transport models for the study area (Hill, 1998; Sutton *et al.*, 1998b). The emissions inventory together with landcover information (to provide dry deposition velocities) was the key input dataset for the multi-trajectory model LADD (Local Area Dispersion and Deposition; Hill, 1998), which was developed at a 50 m resolution. The main aims were to investigate how far the locally emitted NH_3 is transported, and to quantify the high spatial variability of deposition and impacts of NH_3. This is relevant as policies are currently developed to address the impacts of N emissions (Sutton *et al.*, 1998b).

Compared with a national 5 km gridsquare air concentration estimate of 2 $\mu g\ m^{-3}$ for the study area estimated by the FRAME model (Fine Resolution Ammonia Exchange; Singles, 1996; Singles *et al.*, 1998), LADD provided a range of 0.1–85 $\mu g\ m^{-3}$ (Figure 8.5.). The highest air concentrations were estimated for a small part of the study area, while most areas showed much smaller concentrations.

The impacts of local re-deposition are estimated to be largest in the immediate neighbourhood of large point sources such as livestock housing and manure storage facilities, and decline with increasing distance from the source. The rate of decline in concentrations and deposition is approximately exponential and depends on the

The impacts of local re-deposition are estimated to be largest in the immediate neighbourhood of large point sources such as livestock housing and manure storage facilities, and decline with increasing distance from the source. The rate of decline in concentrations and deposition is approximately exponential and depends on the magnitude of the source, background concentrations and prevailing wind directions (Sutton *et al.*, 1998b). For the study area, emissions from the poultry farm influenced concentrations up to a distance of 2.5 km in 1993, whereas a small cattle farm was estimated to enhance deposition within a radius of up to 0.7 km (Sutton *et al.*, 1998b). While low to medium level N deposition onto farmland may not pose a significant problem, air concentrations of the same magnitude may adversely affect sensitive ecosystems in the vicinity of local sources. Areas most at risk are small protected areas (e.g. nature reserves) at the margins of or within intensive agricultural land, where critical loads may be substantially exceeded.

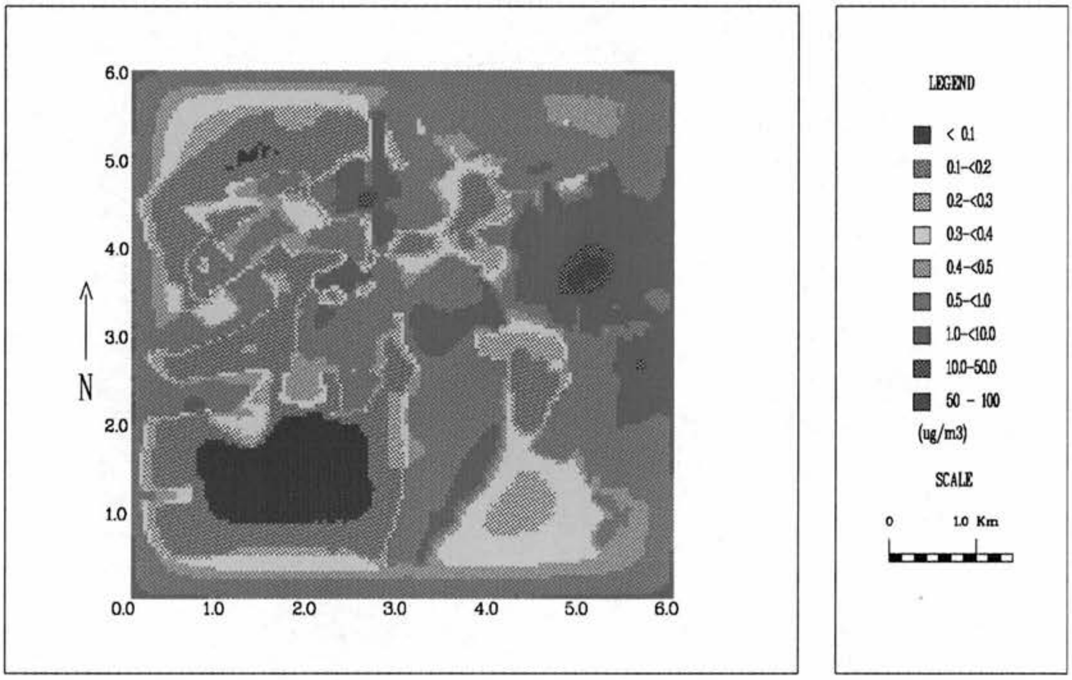


Figure 8.5. Predicted NH_3 air concentrations from the LADD model for the study area, mapped using emissions and landcover information at a 50 m grid resolution. The edge effect is due to the choice of a general background concentration (source: Sutton *et al.*, 1998b)

The case study presented in this chapter highlights the problem of closely interlinked source and sink areas, and the resulting high local variability of NH_3 air concentration and deposition, which leads to significant uncertainties in the accuracy of effects predictions at the national level. Sutton *et al.* (1998b) suggest that current

A local NH_3 emissions inventory for an area in central England was developed to study the variability of emissions at a field scale, as well as to assess the validity of the assumptions regarding agricultural practice in the national inventory. The study area was chosen because detailed information on agricultural practice was available, having been collected with regard to a local agricultural management scheme. Input data were available for 1993 and 1996 on a field by field basis, allowing NH_3 emissions to be calculated for each individual field, separately for livestock grazing, livestock housing and manure storage, landspreading of manures and fertiliser N application to crops and grassland. The study area had to be kept anonymous due to disclosivity issues involved.

The results show that the most extreme local variability in NH_3 emissions is linked to housing and storage losses. However, landspreading of manures and intensive cattle grazing on highly fertilised fields are other important area sources, which vary intimately at the field scale between source and sink areas. Overall, the variability of emissions from agricultural land within the study area ranges from 0–2,000 kg N ha^{-1} $year^{-1}$ in 1993 and 0-8,000 kg N ha^{-1} $year^{-1}$ in 1996, respectively, with the peak at a poultry farm located in the study area. On average, the estimated field level emissions are equivalent to 32.4 kg N ha^{-1} $year^{-1}$over the whole study area.

This compares favourably with the emission for the equivalent 5 km gridsquare in the national inventory for 1996, which is estimated at 34.2 kg NH_3-N ha^{-1} $year^{-1}$ for agricultural sources. Regarding total emissions from fertiliser N application to crops and conserved grassland, good agreement between the average fertiliser application rates applied in the national inventory and the aggregated field data for the study area was found. For individual crop types, the figures matched less closely in some cases, however, on average these discrepancies were evened out over the 5 by 5 km square.

For emissions from livestock, the average estimates from all categories and sub-sources together were again very similar for the national and the local inventory. However, the individual estimates differ widely, due to the different level of detail in the two datasets. A comparison of the results suggests that the national inventory may be underestimating emissions in intensive agricultural areas, where the average source strength assumptions of the UK model are too low. This is especially relevant in areas where large pig and/or poultry enterprises are present. This may be related to two factors: firstly, the emission source strength data for intensive farming such as dairying or intensive beef production are higher than average estimates. Secondly, the redistribution of emission sources from intensive pig and poultry farms over all suitable areas within the source parish may over-smooth emissions from these sources, especially in larger parishes. In more extensively farmed areas dominated by suckler beef and sheep farming, the average emissions estimated by the model may be too high, thus providing overestimates for these areas in the national inventory. It is suggested to assess the stability of the average UK model further by developing detailed local scale inventories for sample areas in other parts of the UK. This approach would also assist in developing and calibrating improved versions of the national model, which take spatially variable emission source strength estimates into account.

The results of this study are also useful to highlight the effects of high local variability in NH_3 emissions on local air concentrations and N deposition, as well as critical loads exceedances. For this purpose, a local scale atmospheric transport model was set up for the study area. Results show that large local sources influence air concentrations in their vicinity considerably and deposition in neighbouring semi-natural areas may exceed critical loads significantly. This emphasises the importance of taking local variability into account when considering the implementation of abatement policies for NH_3.

Chapter 9
Sources of uncertainty in modelling NH_3 emissions over the UK

9.1. INTRODUCTION

9.1.1. Background

Spatially distributed models process large quantities of data from diverse sources and dates, at different scales and with varying levels of uncertainty. The output generated by such models may therefore be affected by many errors and uncertainties introduced during the stages of data processing, from data capture to analysis and presentation. It is important to determine the various sources of uncertainty both in the input data and in the model, as well as to be aware of the implications for the interpretation of the results.

Users and producers of spatial data, particularly in the fields of GIS, remote sensing and environmental modelling, have become increasingly aware of these uncertainties, especially over the last decade or so (e.g. Burrough, 1986; Burrough, 1989; Fisher, 1989; Chrisman, 1991; Congalton, 1994; Prisley, 1994). There is a clear consensus on the importance of uncertainty estimates, whether quantitative or qualitative. This becomes ever more important as the inclusion of spatial analyses in decision making processes "lends an aura of credibility, of careful analysis of alternatives, of superiority of information and knowledge" (Prisley, 1994). It is therefore imperative to minimise uncertainty, while communicating the magnitude, sources and implications of the unavoidable uncertainties which remain in the results.

Numerous attempts have been made at defining and separating the terms error, uncertainty, precision, accuracy, variation, etc., as well as numerous classifications made of error and its sources in spatial data and models (e.g. Burrough, 1986; Collins and Smith, 1994). Burrough (1986) identified three main groups of potential error sources:

- Group I errors ("obvious sources of error") are introduced through the age of the data, the areal coverage, map scale, sampling errors, relevance of the data for the purpose in hand, data format (e.g. vector or raster, classification).
- Group II errors result from natural variations in the phenomena under investigation or from the original measurements. They include the positional accuracy of the data as well as attribute errors (e.g. bias of the observer or malfunctioning of measurement equipment).
- Group III errors arise from the processing of spatial data, such as digitising, interpolation, numerical and logical errors, or methodological problems.

Uncertainties or errors from these three groups are progressively more difficult to detect and to rectify.

In this study, the more generic term uncertainty is used, not only regarding the spatial domain, but all issues relevant to the modelling of NH_3 emissions. This chapter focuses on defining and understanding the different sources of uncertainty encountered in NH_3 emission inventories. The impact of these uncertainties using the model developed here is then considered, together with ways to improve the model output. To this end, sensitivity analyses are presented to give an indication of model reliability. This is achieved by comparing outputs created by using alternative input variables in a systematic and controlled way.

9.1.2. Summary of sources of uncertainty in this study

Kiefer (1994) suggests that the amount and magnitude of uncertainty should always be considered within the context of the application under discussion. Modelling spatially distributed NH_3 emission inventories usually involves integrating source distribution data and emission source strength data. Both these main data sources and the model itself have associated uncertainties, which influence the validity of the resulting emission inventories.

In this study, landcover data and average agricultural practice were used as additional input data to improve the spatial distribution of the model results, compared with the more general approach used by other authors (e.g. Kruse, 1986; Eager, 1992; Sutton *et al.*, 1995). The emission source distribution model presented in this study was tailored specifically for NH_3, not only in an attempt to reduce the spatial uncertainties in the results, but also to explore the sources of uncertainty and their effects on the process of source distribution in more detail.

The uncertainties in the model can be summarised as:

- uncertainties inherent in the spatial input data (parish census data, landcover data)
- uncertainties due to model assumptions regarding emission source strength data (due to environmental factors, agricultural practice and due to differences between different authors' estimates)
- uncertainties due to the model assumptions regarding the spatial distribution of emission sources
- temporal uncertainties (differences between years, seasonality of emissions etc.)

9.2. UNCERTAINTIES IN THE SPATIAL INPUT DATA TO THE MODEL

9.2.1. Uncertainties in the Parish Census data

Data processing techniques, including generalisation and aggregation of the Agricultural Census, contribute to the level of uncertainty in the model output by influencing the reliability of the input variables. There are three main aspects contributing to the uncertainty in the parish census data available for this study: a) the resolution of the parish boundary data, b) the allocation of parish data to county-summary parishes, and c) the allocation of holdings data to parishes.

The parish boundary data have been generalised at the Edinburgh Data Library to a 1 km grid resolution, with the priority to retain the parish area rather than the parish shape. Compared with other input data, the overall uncertainties introduced into the model by the spatial resolution of the parish boundaries, appear to be very small.

There is, however, another aspect of the parish boundary data that may give cause to relatively large spatial displacements of census data for some parishes. The restrictions caused by disclosivity rules and the resulting amalgamation of disclosive parishes to spatially discontinuous county-summary (SDCS) parishes in England &

Wales for 1988 resulted in a potential spatial displacement of emission sources within each county. This is estimated to have led to underestimates of census items and consequently NH_3 emissions for some of these amalgamated parishes, and overestimates for others. For instance, emissions from a large poultry farm in a disclosive parish may be spread over a large numbers of parishes, which were amalgamated for disclosivity reasons. On the one hand, this creates smaller poultry farms in places where there are none in reality, while also hiding or flattening the real hot spot at the other side of the county.

For the 1996 inventory, this uncertainty was minimised by the availability of disclosive parish data. These were treated according to the rules outlined in an agreement with MAFF and SOAEFD, i.e. potentially disclosive parishes were amalgamated with immediate neighbours rather than all other potentially disclosive parishes in the same county. Thus, emission sources originating in a potentially disclosive parish are still located relatively close to the original source, and are diluted over a smaller area.

The effect of this improvement on the spatial distribution of the census data was quantified through a sensitivity analysis. For this purpose, SDCS parishes were created for 1996 as described above for 1988. This enabled a comparison between emission inventories created using two different methodologies for treating disclosive parish data for the same year (1996):

(a) amalgamation of disclosive parishes with neighbouring parishes, until non-disclosive output at the 5 km grid level is guaranteed (optimal solution as used in the model for 1996 presented here)

(b) amalgamation of all disclosive parishes within each county into one SDCS parish (methodology used by MAFF for 1988 parish census data).

The results of this sensitivity analysis are shown in Figure 9.1.(a, b) for total livestock emissions in eastern England. Firstly, the map created using the optimal methodology (Figure 9.1a) has a larger number of 'hot spots', while some of these were diluted significantly through the creation of SDCS parishes (Figure 9.1b). This leads to areas with low emissions showing increased emissions in Figure 9.1b.

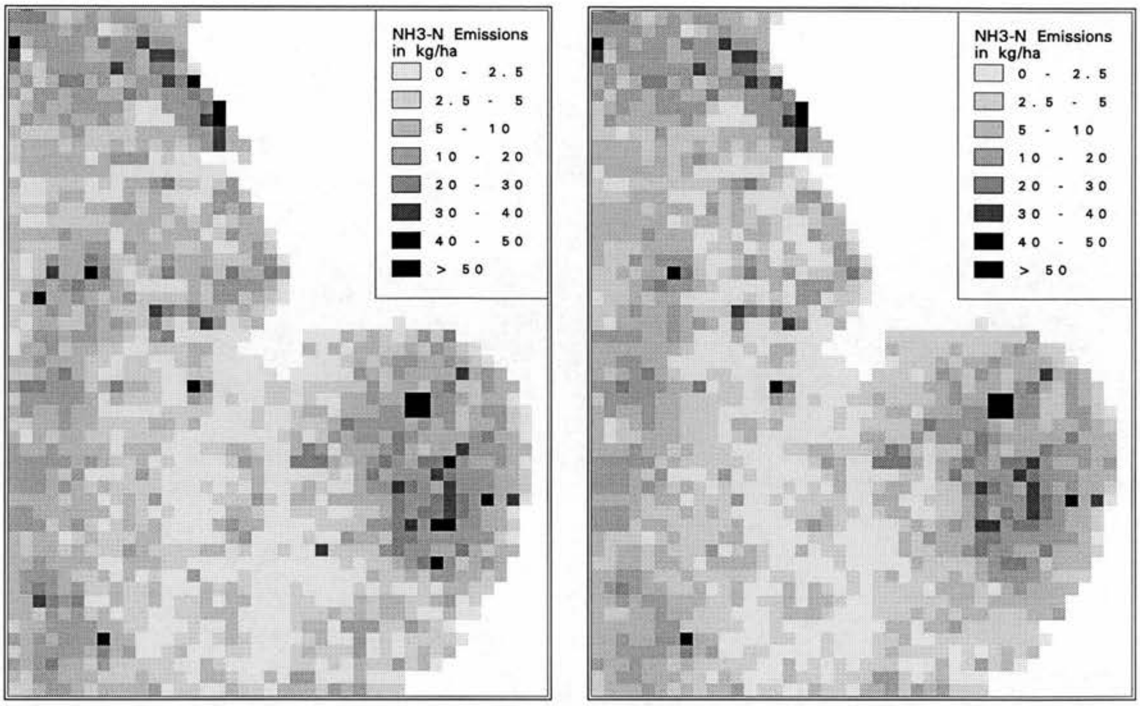


Figure 9.1. Ammonia emissions in eastern England in 1996, modelled (a) using an optimal method for the amalgamation of disclosive parishes (amalgamation with neighbouring parish(es) as required), and (b) amalgamating all disclosive parishes for each county into one SDCS parish.

Figure 9.1. (a, b) and Table 9.1. highlight the differences between the 2 methodologies employed on a per-gridcell basis. Table 9.1. further quantifies how emission 'hot spots' are smoothed out considerably, i.e. emissions are by up to 60.6 kg N ha^{-1} per gridsquare smaller, in the SDCS approach. In squares with positive differences in Table 9.1., where in the SDCS approach gives larger values than the optimal approach, emissions are overestimated by up to 15.1 kg N ha^{-1}. Although the maximum increase in emissions in the SDCS approach is smaller than the maximum decrease, emissions are overestimated by more than 1 kg N ha^{-1} in ~15% of all squares through the county-wide dilution of emission hot spots from disclosive parishes. Hence not only do emission hot spots become diluted, but emissions are also overestimated significantly in the areas to which the original hot spots are redistributed. It should also be noted that the effects of using an SDCS approach as in Figure 9.1.b would be even more pronounced, if disclosivity rules had been applied separately for all livestock and crop categories, rather than aggregated to all livestock emissions and all crop emissions.

Table 9.1. Difference between emissions on a per-gridcell basis, modelled (a) using an optimal method for the amalgamation of disclosive parishes (amalgamation with neighbouring parish(es) as required, and (b) amalgamating all disclosive parishes for each county into one SDCS parish; in eastern England in 1996. Negative values indicate that the emission estimates are larger in the associated gridsquares for the optimal method than for the SDCS method (i.e. 'hot spots' are smoothed out in the SDCS map). Positive values indicate increased emission estimates when using the SDCS method as compared with the optimal method.

Difference in emission per gridcell (kg N ha^{-1} $year^{-1}$)	% of gridsquares
-100 to –50	0.1%
-50 to –30	0.1%
-30 to –20	0.3%
-20 to –10	0.7%
-10 to –5	2.1%
-5 to –1	14.3%
-1 to +1	67.9%
+1 to +5	13.6%
+5 to +10	0.8%
+10 to +20	0.2%

The third major uncertainty inherent in the parish census data is due to the allocation of holdings data to parishes. It has already been described in detail, how the amalgamation of holdings data to civil parish level may cause the spatial mis-location of a holding in a neighbouring parish (Hotson, 1988; see Figure 4.3.; Section 4.2.1.). This problem can be attributed to the fact that the combined area of all the holdings contributing to the census returns for a civil parish is not necessarily congruent with the area of the civil parish, i.e. most of the time farm boundaries do not coincide with parish boundaries. Any farm may thus be counted with one parish for census purposes, but have much of its agricultural activity in one or several neighbouring parishes. This can not be circumvented as long as the spatial reference of the model input data is a parish membership of the farm address. Checks can be carried out for crops by comparing areas of crops in the census data and potential locations for them on the landcover map. For grazing livestock (cattle, sheep, goats, deer and horses), stocking densities can be checked after redistribution within the parishes. In most cases this mis-location does not pose too severe a problem (J.McG. Hotson, pers. comm., 1998).

In the literature, the issue of how the choice of areal units affects the spatial aggregation of single observations, is described as the 'Modifiable Areal Unit Problem' or MAUP (Openshaw 1984; Dudley 1991; Fotheringham and Wong 1991). Amrhein and Griffith (1994) divide the effects of the aggregation of spatially located data points into two main aspects: the 'scale effect' and the 'zonation effect': This introduces further uncertainties in the census data:

a) The *scale effect* (see difference between Figure 9.2a and 9.2b) refers to the number of areal units created through amalgamation, which influences the usefulness of the aggregated data at certain resolutions. In this study, this problem is represented by the parish size. The larger the parish, the more diluted any individual features within the parish become. In the model presented in this thesis, one aim was to disaggregate the parish census data to a probable spatial distribution within each parish, dependent on conditions linked to landcover data. Although uncertainty is introduced through the aggregation of single data points to non-congruent areal units, and also through the disaggregation process presented in this study, the end result has been shown to be more realistic than without the spatial disaggregation model (Chapters 5, 6).

b) The *zonation effect* (see difference between Figures 9.2b and 9.2c) refers to the allocation of the single observations to the areal units. For instance, if the holdings data collected for the Agricultural Census were amalgamated to a different set of areal units (e.g. postcodes or enumeration districts), the total sum of the different census items present within each areal unit would result in a different spatial distribution of NH_3 sources within a relatively local neighbourhood around each single observation (e.g. all the new areal units overlapping with the parishes in the present amalgamation). In the present study, this is not just due to the normal aggregation effects resulting from the data generalisation, but also due to the mismatch between the area occupied by the civil parishes and the sum of the area of the holdings allocated to these parishes.

Another example for the scale effect (and to a lesser extent also for the zonation effect) is the sensitivity analysis described above, regarding the treatment of disclosive parishes. The creation of county-wide summary parishes introduces larger parishes, which result in a smoothing of the spatial distribution of NH_3 sources. Compared with this, the optimal approach keeps the size of amalgamated parishes as small as possible and the data allocated to parishes in a spatially continuous area.

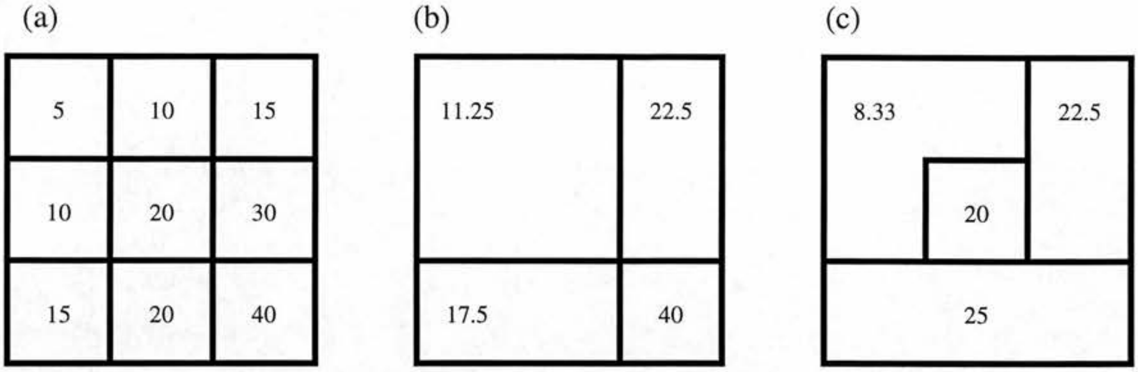


Figure 9.2. Scale (compare (a) and (b)) and zonation effects (compare (b) and (c)) caused by the aggregation of individual data points. After Amrhein and Griffith, 1994.

It is important here to keep the origin and format of the input data and the processing history of the model results in mind. One approach taken here to minimise this problem is the aggregation of the model results from a 1 km processing level to a 5 km publication level. This helps to avoid the mistake of treating the results too literally and believing the numerical precision to represent the accuracy of the redistributed output at a large scale. For instance, emission sources originating from a large poultry farm, which is located in a relatively small area of a large parish, will be dis-aggregated in the model over all potentially suitable land within the parish, and thus the subsequent NH_3 concentrations may be diluted significantly, compared with reality. Additionally, the poultry farm may be registered in one parish, but some of its poultry houses or landspreading areas may be located in a neighbouring parish.

9.2.2. Uncertainties in the landcover data

Estimating the accuracy of landcover maps derived from satellite images has been an integral part of landcover classification and a focus of research in the remote sensing community (Congalton, 1991; Fuller *et al.*, 1994). For most classified satellite landcover data sets, estimates of errors and uncertainties are well documented and allow users to assess the overall data quality. In general, the accuracy of satellite classifications depends on many different aspects, such as:

- the size of the landcover features compared with the spatial resolution of the sensor(s) - causing mixed pixel effects,
- the terrain surveyed (slope, aspect, etc.),
- the interference by the earth's atmosphere (cloud cover, haze etc.),

- the suitability of the wavelength channel(s) of the sensor(s) for distinguishing the features on the earth's surface,
- the classification method(s) applied,
- the data available for ground truthing,
- the time of year when the data are collected.

The most commonly used method of representing the degree of accuracy of a classified image is a confusion (also error or agreement) matrix (Tables 9.2., 9.3.). For the landcover data used in this thesis (ITE landcover map, ILC90), Fuller *et al.* (1994) compared field reference data from the Countryside Survey 1990 (CS90) (Barr *et al.* 1993) for 143 1-km squares with their classified results (derived from a mosaic of multi-seasonal satellite images from 1990 ± 2 years). It has to be emphasised that the reference surveys were undertaken on different dates from the satellite images, for a different purpose and thus with a different classification. Therefore comparisons such as given in Table 9.2. can only give estimates regarding the accuracy of the classified landcover data, which will nonetheless alert the user to potential sources of uncertainty.

According to Fuller *et al.* (1994), a large proportion of the discrepancies between the 2 datasets is due to significant differences in class definitions, mainly for bogs and the continuum from managed grass via heather/grass to shrub heaths. This is especially relevant for similar categories, such as rough grass and heather/moor grass, which are in many cases far from discrete, easily distinguishable cover types. Allowing for differences in definition as well as the fact that e.g. managed grassland within the suburban/urban areas were ignored in the field survey, Fuller *et al.* (1994) estimate the correspondence between the 2 datasets at 67%. When temporal changes between the two surveys (sometimes up to 2 years time difference), such as ploughing of pastures, are taken into account, the overall correspondence percentage rises to 76%. Fuller *et al.* (1994) assign the main component of uncertainty to the effects of misclassifications of mixed pixels, mostly at boundaries between different features.

Comparisons between the ITE Landcover Map and other landcover datasets for the UK, e.g. the Landcover of Scotland data (LCS88), which was derived from aerial photographs, have also been undertaken (Brooker, 1995). Brooker describes substantial disagreement between the 2 datasets, for some classes more than for others. However, a large proportion of this is again due to the different classifications used. For instance, Table 9.3. distinguishes only one aggregated class for coastal features in each dataset, which includes coastal bare ground, saltmarshes and sea/estuary, rather than the 3 classes in Brooker's original table. This simple exercise improved the correspondence between the two datasets for coastal features significantly, by evening out differences which can mainly be attributed to the state of the tides, rather than any real mis-classifications.

Other reasons for differences between the two datasets can be found in the nature of the data collection methods and the classification methods applied to them. The satellite data were classified from multi-seasonal and multi-channel images, which is advantageous when tilled land is to be distinguished from (permanent) grassland. This differentiation is difficult with single-season aerial photos, as cereal fields at a certain stage of growth are very difficult to distinguish from grassland. On the other hand, features such as urban and suburban grassland (e.g. sports facilities or parks) are easily distinguished from agriculturally used grassland on (manually classified) aerial photographs, as the interpreter can classify in context, whereas both categories show very similar spectral characteristics in automated or semi-automated per-pixel satellite image classifications. Temporal discrepancies between the two datasets (up to 4 years), are likely to account for some of the confusion between tilled land and managed grassland due to crop rotation.

Despite the uncertainties at the large scale, for the final resolution required for the NH_3 emission model, i.e. 5 km grid squares, there is a generally good fit between different landcover data sources for the relevant aggregated categories, and the differences and uncertainties can mostly be neglected. Regarding the spatial distribution of NH_3 emissions, either dataset would be suitable as model input, as the main purpose is to differentiate between intensively and extensively used areas.

Table 9.2. Confusion matrix: Correspondence between CS90 Field Survey and ILC90 for 143 1-km squares, measured on a 25 m grid (after Fuller *et al.* 1994): values in percent.

ILC90 / CS90	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)	(14)	(15)	(16)	(17)	**Total %**
Unclassified (1)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**.**
Suburban/urban (2)	.	**3**	.	.	.	.	.	2	2	.	.	.	.	.	.	.	.	**8**
Sea/estuary (3)	.	.	**2**	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**2**
Inland water (4)	.	.	.	**1**	.	.	.	.	.	.	.	.	.	.	.	.	.	**2**
Coastal bare (5)	.	.	1	.	**1**	.	.	.	.	.	.	.	.	.	.	.	.	**3**
Saltmarsh (6)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**0**
Inland bare (7)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**1**
Tilled land (8)	1	1	.	.	.	.	.	**14**	3	.	.	.	.	.	.	.	.	**21**
Managed grass (9)	1	.	.	.	.	.	.	3	**18**	1	.	2	.	2	.	1	.	**29**
Rough/marsh grass (10)	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	**2**
Bracken (11)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**1**
Heath/moor grass (12)	.	.	.	.	.	.	.	.	.	.	.	**2**	.	2	.	.	.	**6**
Bog (13)	.	.	.	.	.	.	.	.	.	.	.	2	**1**	5	1	.	.	**10**
Open shrub (14)	.	.	.	.	.	.	.	.	.	.	.	1	.	**2**	1	.	.	**5**
Dense shrub (15)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	**1**
Deciduous/mixed (16)	.	.	.	.	.	.	.	1	1	.	.	.	.	.	.	**2**	.	**6**
Coniferous (17)	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	1	**1**	**4**
Field-surveyed	**3**	**5**	**3**	**1**	**2**	**0**	**1**	**21**	**27**	**2**	**2**	**9**	**3**	**13**	**3**	**4**	**2**	**100**

direct correspondence (sum of diagonal)	46%
correspondence allowing for differences in cover-interpretation between surveys	67%
correspondence allowing for differences in cover-interpretation and changes in time between surveys	76%

Table 9.3. Confusion matrix between LCS88/ILC90: number of 25 m cells in each category (modified after Brooker 1995)

ILC90 categories / LCS categories	Tilled land	Bogs (A)	Built-up land (A)	Coniferous/ evergreen woodland	Heather moor (A)	Felled wood-Land	Managed grassland (A)	Grass heath (A)	Bracken	Deciduous woodland	Scrub/ orchard	Inland water	Rough/ marsh grass	Coastal (A)	SUM (row)	**% of SUM in diagonal**
Arable	**139,492**	3,801	2,026	887	2,356	0	41,283	4,502	6,419	4,661	0	7	2,887	17	208,338	**67.0**
Blanket bog/peat bog	536	**11,723**	45	563	69,947	27	4,530	17,853	581	1,349	0	304	4,271	18	111,747	**10.5**
Built up (A)	4,387	246	**6,650**	82	4,601	0	2,083	599	287	238	0	8	1,164	41	20,386	**32.6**
Conifer woodland (A)	3,590	4,029	174	**95,274**	21,265	3	4,307	5,632	298	8,201	0	112	6,733	0	149,618	**63.7**
Heather moor (A)	3,210	11,297	131	3,971	**435,970**	81	13,172	30,582	2,947	6,781	0	372	1,921	426	510,861	**85.3**
Felled woodland (A)	2,727	5,499	265	2,829	24,687	**7**	5,885	18,107	1,570	1,804	0	47	8,632	0	72,059	**0.0**
Managed grassland (A)	27,946	1,670	600	2,974	25,987	0	**266,523**	30,280	11,405	10,779	0	398	23,825	546	402,933	**66.1**
Grass heath (A)	29	493	23	333	5,068	0	3,792	**11,256**	379	505	0	6	1,549	0	23,433	**48.0**
Bracken	333	78	0	294	5,089	0	13,367	4,521	**2,035**	1,468	0	0	1,527	0	28,712	**7.1**
Broadleaf	1,352	227	79	1,239	3,958	0	2,509	867	158	**3,939**	0	44	1,081	28	15,481	**25.4**
Low scrub	37	7	0	7	51	0	323	37	15	51	**0**	0	59	0	587	**0.0**
Water	448	802	95	737	3,055	0	342	673	15	597	19	**16,500**	197	4	23,484	**70.3**
Wetlands	2,236	688	43	1,505	6,579	0	19,006	9,718	645	5,633	0	1,204	**7,224**	129	54,610	**13.2**
Coastal (A)	357	541	180	49	4,503	0	4,460	4,613	30	236	0	38	2,000	**15,147**	32,154	**47.1**
SUM	186,680	41,101	10,311	110,744	613,116	118	381,582	139,240	26,784	46,242	19	19,040	63,070	16,356	**1,654,403**	
% of SUM in diagonal	**74.7**	**28.5**	**64.5**	**86.0**	**71.1**	**5.9**	**69.8**	**8.1**	**7.6**	**8.5**	**0.0**	**86.7**	**11.5**	**92.6**		

(A) .... aggregation of at least 2 original classes

9.3. UNCERTAINTIES IN THE EMISSION SOURCE STRENGTH DATA

Ammonia emission source strength per source unit varies over the country depending on both environmental factors and farming practice. In the basic model developed for this study (Chapters 4-7), average values were used over the whole of the UK, without taking into account the variability caused by environmental factors and farming practice.

For example, the maximum length of the grazing period for cattle varies considerably in the UK. This in turn determines the proportion of NH_3 volatilised per animal, as the time animals spend indoors is associated with higher emission rates than the time spent on pasture (see Chapter 10). This relationship between NH_3 source strength and the grazing period, which is governed by climatic factors, is complicated by local farming practice, which may circumvent environmental limitations by extending intensive practices throughout the year. For instance, farmers may rear their young beef cattle entirely indoors.

Another example of variability of emission source strength is due to the different amount of fertiliser applied to the same crops by different farmers. On the one hand this is a random factor dependent on the particular farmer's situation and his attitudes towards recommendations by agricultural advisory services, organic farming, etc. On the other hand, environmental factors such as climate, topography and soil conditions play a role in what crop and grass yields are economically achievable with optimal fertiliser application rates (see Chapter 2), and this would in turn influence the associated NH_3 emission potential (Chapter 10).

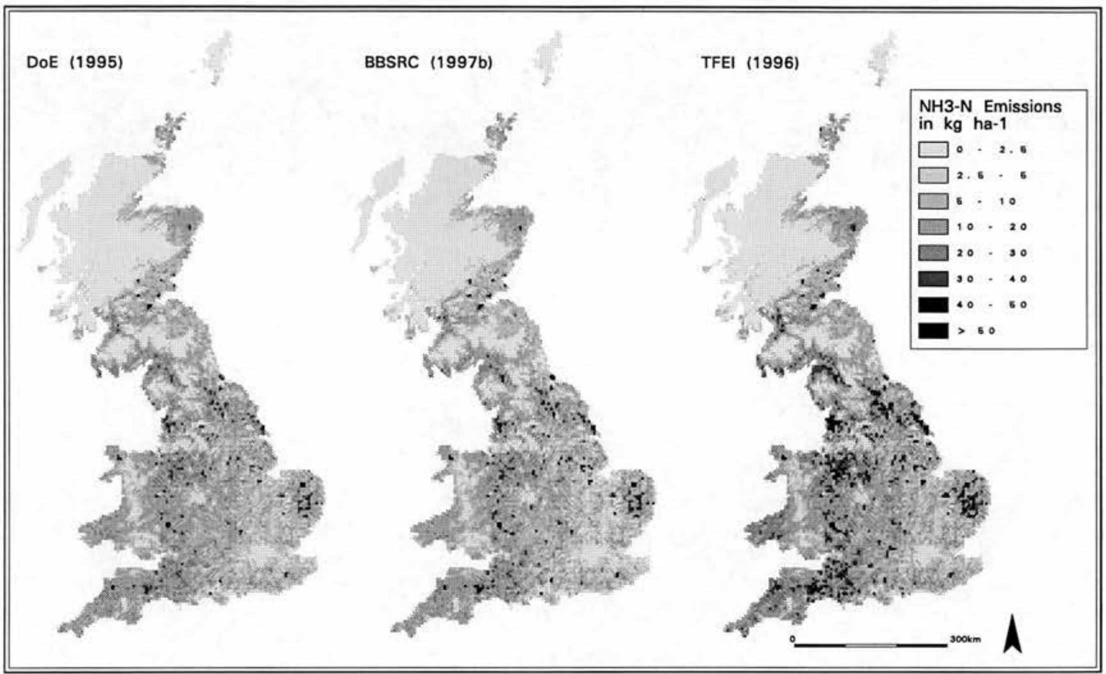
The uncertainties caused by the random effects of individual farming practice, as discussed above, cannot be removed from the model results without the use of specific data for each farm. It is, however, possible to investigate this type of uncertainty in more detail for sample areas, such as the field scale study in Chapter 8. In general, the random effects of individual farming practice on small and medium sized farms are expected to be more or less balanced out within the 5 km grid resolution of the national inventory (Table 8.6.). Large enterprises such as intensive pig or poultry farms, however, may cause significant over- or underestimates in NH_3 emissions, depending on the farming practice employed. A register of the largest enterprises as suggested by IPPC (EC, 1996), which could include basic management information, would be extremely useful for reducing the significant uncertainties linked with these largest and - regarding N deposition and its effects - most critical NH_3 sources.

The only way of addressing the problem of random deviations from the average at all levels would be by farmers waiving their right to anonymity, so that their complete census return could be used in any inventory. However it is likely that in the UK this would be politically unacceptable, although it has been done elsewhere in Europe, e.g. in Switzerland (Rihm, 1994). Such an operation would also address the less random deviations due to the length of the grazing season and soil fertility. In the absence of such detailed information these less random deviations may be modelled to investigate their influence on NH_3 source strength (Chapter 10).

Deviations from average conditions are not the only cause of uncertainty regarding emission source strength estimates in NH_3 emission inventories. As discussed in Chapter 3, differences in opinion between the scientists who have developed recent inventories for the UK regarding the average emission source strength estimates, especially for different agricultural livestock types, provide one of the largest uncertainties in the model. While the basic model introduced in this thesis uses a set of source strength data agreed by MAFF and the main scientific groups involved in NH_3 emissions in 1995 (DoE, 1995; RGAR, 1997) for submission to EMEP, these have recently been superseded by the latest BBSRC inventory (BBSRC, 1997b). Other possible source strength estimates for application in the model are Sutton *et al.* (1995) and the CORINAIR/UNECE inventory (TFEI, 1996). The latter is considered here as the current best estimate of emission source strength (Chapter 3).

A sensitivity analysis was carried out to show the effects of using different emission source strength data. For this purpose, the model output versions created by using 3 different sets of source strength data (DoE, 1995; BBSRC, 1997b; TFEI, 1996) were compared with each other. This resulted not only in different total emissions, but also in shifts in the importance of some sources compared with others, together with different spatial patterns (see Section 3.1. and Figure 9.3.).

In absolute terms, emissions from all livestock sources, separately and aggregated, are larger in the TFEI inventory than in the DoE (1995) inventory (Tables 3.1., 9.4.). Emission estimates from the BBSRC inventory (BBSRC, 1997b) are smaller than the DoE (1995) estimates for cattle and sheep, and larger for pigs and poultry (compare Table 3.1; Table 9.4.).



Figures 9.3. Ammonia emissions from agricultural sources for 1996 in kg NH_3-N ha^{-1}: a) DoE 1995 inventory, b) BBSRC 1997b inventory; c) TFEI 1996 inventory, using the new source distribution model.

Table 9.4. Comparison of emissions (in kt NH_3-N $year^{-1}$) from agricultural sources in the UK in 1996 for the inventories by DoE (1995), TFEI (1996) and BBSRC (1997b).

	Largest estimate	Middle estimate	Smallest estimate
Cattle	TFEI (178)	DoE (134)	BBSRC (126)
Sheep	TFEI (23)	DoE (16)	BBSRC (13)
Pigs	TFEI (30)	BBSRC (26)	DoE (24)
Poultry	BBSRC (44)	TFEI (43)	DoE (27)
Total livestock	TFEI (268)	BBSRC (209)	DoE (207)
Fertilisers	DoE (31)	TFEI (25)	BBSRC (18)
Total agriculture	TFEI (298)	DoE (233)	BBSRC (226)

The total agricultural emissions are 28% larger in the TFEI (1996) than in the DoE (1995) inventory, and 3% smaller in the BBSRC (1997b) inventory than the DoE (1995) emissions. An analysis of the spatial distribution of emissions from the three inventories (Figure 9.3.) shows a more complex pattern. Cattle and sheep dominated areas (compare Figure 6.11.) show a substantial increase in emissions in the TFEI inventory, compared with the other two maps. Similarly, a smaller increase in the TFEI map is visible in Figure 9.2. for pig dominated areas. The poultry dominated areas are least conspicuous in the DoE (1995) inventory. They are most prominent on the BBSRC (1997b) map, which has the largest total emissions of the 3 inventories from poultry, and the smallest for all other source categories except pigs (Table 9.4.).

A simple statistical analysis of the maps was carried out to determine differences on a gridsquare basis (Figure 9.4.). The BBSRC (1997b) inventory is characterised by the highest proportion of gridsquares in the lower categories (<10 kg N ha^{-1} $year^{-1}$). In the highest categories (>20 kt N ha^{-1} $year^{-1}$), however, it has a marginally higher number of high emission gridsquares than the DoE inventory. This can be attributed to the higher source strength estimates for poultry. The TFEI (1996) map contains a larger proportion of squares with higher emissions (> 20 kg N ha^{-1} $year^{-1}$), which may be associated with higher estimates for most livestock emission sources, especially cattle.

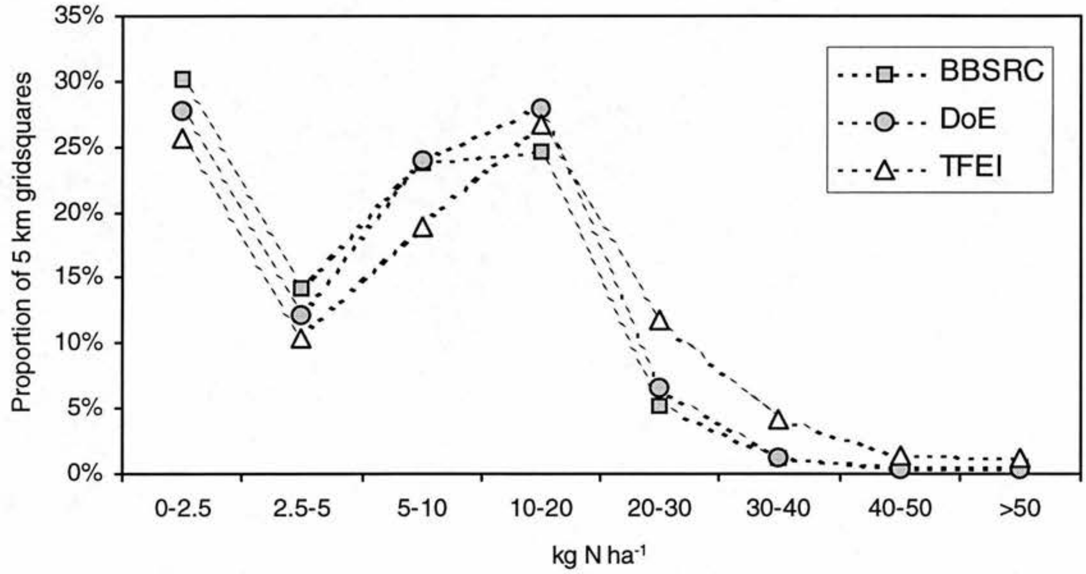


Figure 9.4. Proportion of 5 km gridsquares in NH_3 emission categories in 3 emission inventories (DoE, 1995; TFEI, 1996; BBSRC, 1997b), analysed for total agricultural emissions in Great Britain in 1996 (NB: The data points were joined up to improve the readability of this graph) .

In order to resolve some of the major uncertainties in the emission source strength estimates as a core model input, further research is needed. It appears that one of the main sources of disagreement regarding this problem is the amount of N excreted by the different livestock types (see Chapter 3).

9.4. UNCERTAINTIES IN THE SPATIAL DISTRIBUTION OF EMISSION SOURCES

Uncertainties are also introduced by the modelling process itself. The rules for the redistribution of census items over the landcover data, e.g. stocking densities for grazing, are designed to match average farming practice, not taking any regional differences into account. For instance, a higher proportion of manure has to be spread on grassland in areas with limited areas under arable crop production.

At present, all emission sources originating in one parish are redistributed within the same parish. In some instances, for example the spatial distribution of landspreading of wastes from large intensive livestock farming developments (pigs or poultry), this may cause unrealistically large emissions within the boundaries of the concerned parishes. In reality, livestock wastes from such developments are often spread over a much larger area, landfilled or incinerated in so-called 'poultry powerstations' (e.g. near Thetford, East Anglia).

It is helpful to consider the spatial uncertainties within each 5 km grid area estimated by the present model. Since the 5 km estimates are mean NH_3 emissions, which are actually calculated at a 1 km level (n = 25), it is straightforward to show other statistics for each 5 km grid cell. The variance of NH_3 emissions from agricultural sources for Great Britain in 1996 is shown in Figure 9.5. This map shows the aggregated deviation of the 1 km grid cells from the average value represented for the 5 km cell. It highlights mainly the areas with large pig and poultry farms (compare Figures 6.2c and d), which are also more difficult to spatially distribute in the first place.

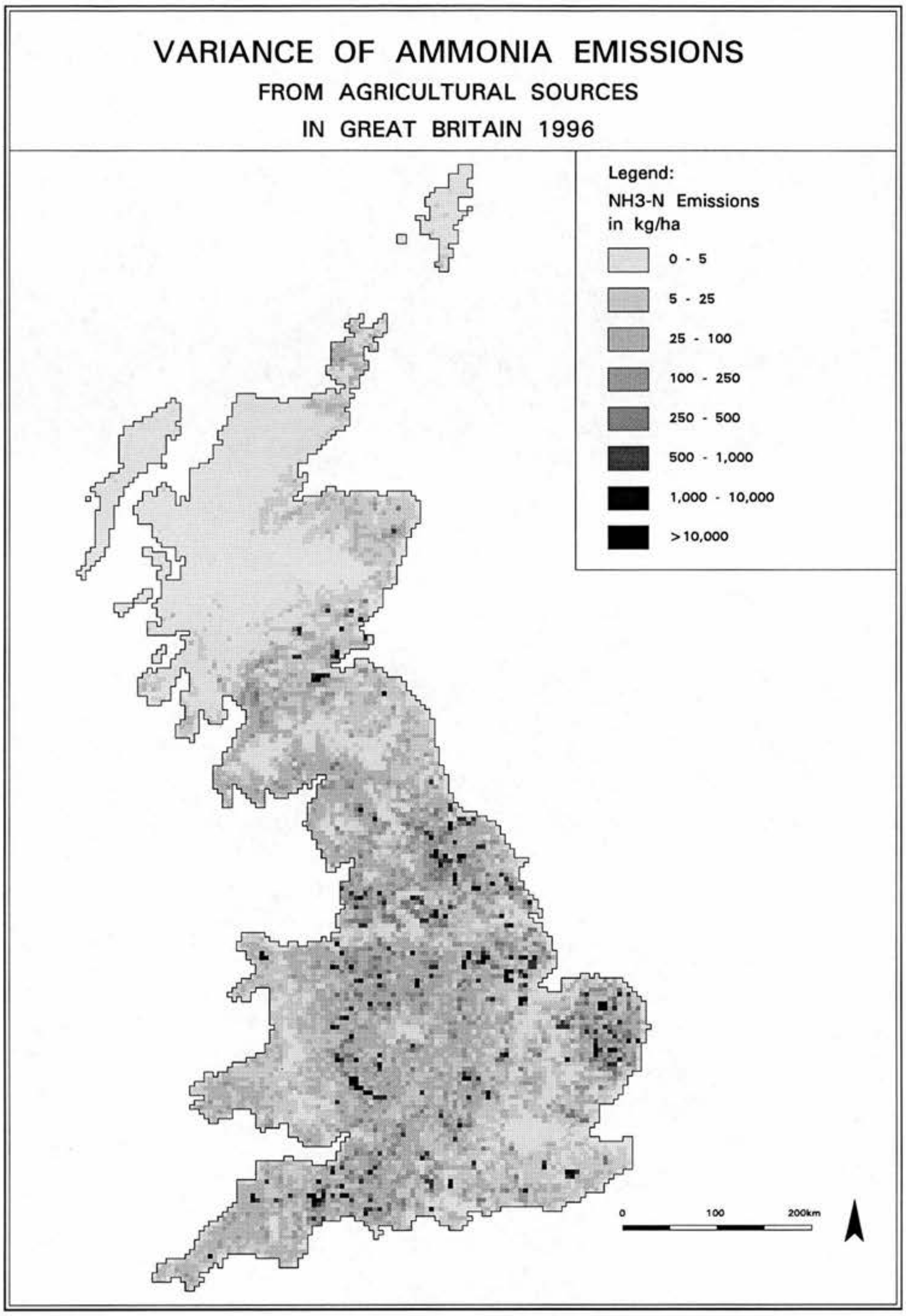


Figure 9.5. Variance of agricultural NH_3 emissions for Great Britain in 1996.

Another example for this is considered in Figure 9.6., which shows the % coefficient of variation of the 1 km estimates (standard deviation / mean * 100). The interpretation of this map may not be immediately obvious. However, a closer inspection shows that the areas of high variability are a) primarily upland areas with an intimate mix of hills and intensively farmed valleys, and b) intensively farmed areas with a mixture of especially pig and poultry emissions.

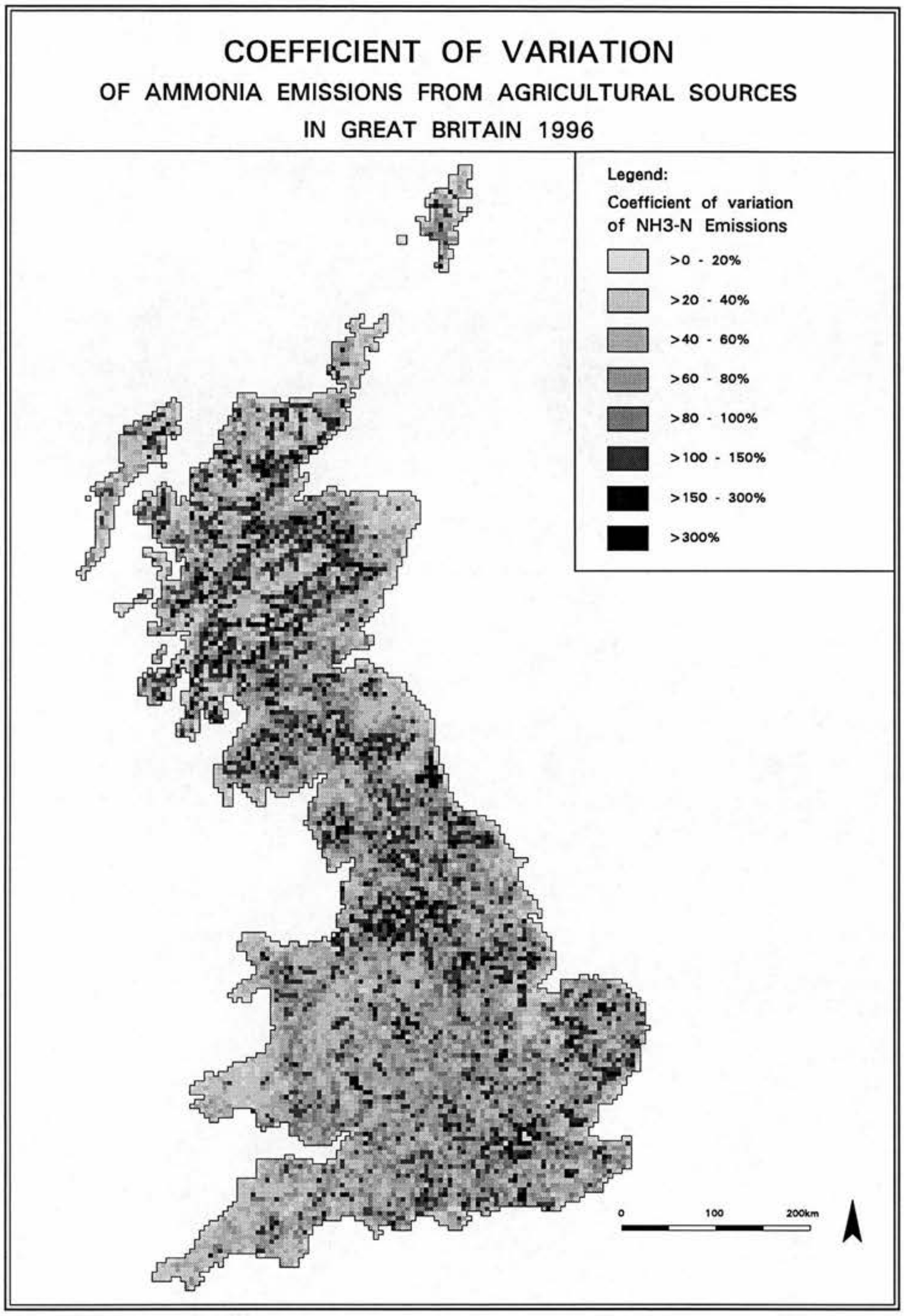


Figure 9.6. Coefficient of variation of NH_3 emissions for Great Britain in 1996.

The lowest values can be found in lowland areas dominated by cattle and sheep emissions, where grazing forms a major part of the agricultural landscape. Thus a high coefficient of variation may be seen in the Lake District and Snowdonia (hills with intensively farmed valleys between them) as well as in East Anglia (intensive pig and poultry emissions). The example of the Highlands of Scotland is interesting, where high uncertainties are located near the boundaries of agricultural areas, and much smaller uncertainties in remote areas, where there is little intensive agriculture even in valleys.

A map such as shown in Figure 9.6. includes both variability that is real due to local topography etc. and variability due to model uncertainties. However, it illustrates the potential at local level for 5 km maps hiding much of the variability. This leads to both areas for which emissions are underestimated (intensive agricultural land) and areas where emissions are overestimated (hill areas, semi-natural land) within the 5 km squares. Such features argue for the continued development of methods to improve the spatial resolution of NH_3 emission estimates.

When viewed together with the emission maps, Figures 9.5. and 9.6. provide additional insights in the spatial structure of NH_3 emissions. Thus they can assist in determining areas with a high local variability, which is hidden by the 5 km output resolution. This indicates areas with higher uncertainty, when the modelled average values at the 5 km grid resolution are used to predict air concentrations with atmospheric transport models, or are compared with air concentration measurements at any location within the gridsquares.

9.5. TEMPORAL UNCERTAINTIES

While it has so far been implicitly assumed that annual emissions from a specified NH_3 source would be the same for two consecutive years, this is by no means true. With the same number of animals on the same farm under the same feeding, housing, manure storage and spreading regime, the resulting NH_3 emissions vary due to variations in meteorological conditions, such as temperature, precipitation, turbulence etc. Under warmer conditions, for instance, volatilisation rates would generally rise, due to reduced solubility of NH_3 in surface water films. This variation between years is not only significant regarding emissions, but also affects atmospheric transport, deposition and effects on sink areas.

Emission rates not only vary between years, but also at a finer temporal resolution such as seasons, months, days and hours. This is not only due to meteorological conditions such as temperature or rainfall, but also due to agricultural activities, which are strongly linked to the natural growing cycle as well as a diurnal cycle

(Kruse *et al.* 1989, Asman 1992b). Some activities such as manure spreading or fertiliser application give rise to very high short-term or acute peaks in emission which tail off quickly, while other sources such as animal housing, grazing or manure storage provide more long-term chronic emissions. Some activities are linked to certain seasons, such as fertiliser or manure application to crops and livestock grazing, while others may occur at any time of the year such as poultry or pig housing.

Thus, the annual average emissions calculated in the model may not be different to the emission events occurring at a particular location during a particular period of time. For instance, while manure spreading emissions may be moderate in size when aggregated over a whole year, very high emission peaks will dominate over other sources during the first few hours or days after the spreading event. These circumstances have to be taken into account especially when the final objective of NH_3 emissions modelling is effects-based abatement, as vegetation is sometimes considered more susceptible to N deposition during the growing season (Asman, 1992b) than at other times of the year.

Seasonal emission inventories would be of considerable importance for improving atmospheric budget calculations, as there may be non-linearities coupling between emissions atmospheric transport and deposition. Splitting the annual emissions into seasonal or monthly parts for input to transport and deposition models, based on average meteorological conditions, can also contribute to improvements regarding uncertainties in the modelled distribution of effects. This may be attributed to significant changes in weather patterns, especially wind speed or prevailing wind directions during the course of a year. Such a procedure would realistically need to address temporal variability with much more detailed meteorological data, in order to avoid the pitfall of increasing precision whilst reducing the accuracy. Considering emissions via atmospheric transport and deposition to effects as a continuum, it is even more important to keep these sub-annual variations and variabilities in mind, when interpreting the results.

9.6. DISCUSSION AND CONCLUSIONS

Critical assessment of models includes an investigation of the uncertainties in the resulting output. There is an increasing awareness of this issue among model developers and users, driven by the need to estimate the reliability of the end product. This is especially important as emission models may be used for decision-support regarding abatement of pollution etc. Qualitative and quantitative statements about the uncertainties involved also give indications of the sensitivity of the model to certain input parameters and thus aid the interpretation of the results. The sensitivity of a model may be estimated by comparing model outputs from different scenarios, i.e. by using alternative input variables or making changes to the model in a systematic and controlled way.

In this chapter, a brief literature review on uncertainties in spatial modelling is presented, and the main sources of uncertainty encountered with the model developed in this thesis are discussed. The following groups of uncertainties are addressed here:

- *Uncertainties due to the spatial datasets used as model input*

  In general it can be stated that both the parish census data and the landcover data provided for this study are suitable and adequate for the purpose of a national emission inventory at the given spatial resolution of 5 km gridcells. However, several problems have been identified with the parish census data. They are mostly related to smoothing effects due to the aggregation of spatially located data points (MAUP). For the model presented here, they are caused by the need to use variable size parishes and parish groups as the base unit, rather than individual spatially located holdings. This approach was necessary due to restrictions imposed by MAFF and SOAEFD, to ensure non-disclosivity of the census data at the output level.

- *Uncertainties due to model assumptions regarding source strength data*

  Several possible sets of average source strength data were used in the model, while keeping all other parameters unchanged. This provided an opportunity for sensitivity tests regarding the uncertainty in the magnitude of emissions. It is shown that the general spatial pattern of the different scenarios is stable, despite changes in the importance of some source sectors relative to others, and overall increases in the magnitude of emissions with higher source strength data.

A second set of uncertainties has been identified regarding the use of the same average source strength estimates all over the UK, without taking spatial variations over the country into account. These variations may be attributed to two groups, systematic variations (related to regional trends in environmental factors and agricultural practice over the country), and random variations (mainly due to individual behaviour of the farmers). The former are discussed in more detail in Chapter 10, where suggestions for future work are outlined. In order to provide a deeper insight into the significance of random variations in emission source strength, permission to use data at the holdings level would be required.

- *Uncertainties due to model assumptions regarding spatial distribution of emission sources*

  Two main sources of uncertainty have been identified here:

  a) Firstly, the rules governing the spatial distribution of emission sources within the model, e.g. relative differences in stocking densities for different quality grazing land, were designed to match average farming practice, thus not catering for any variations between farmers. This is, however, a limitation that could only be fully resolved with detailed management and locational information for every holding. For the given spatial resolution of the inventory at the 5 km grid level, this amount of detail is not only unnecessary, but also detrimental to efficient data processing in the model.

  b) Secondly, the model redistributes emission sources from agriculture only within the parish of origin. This is likely to cause overestimates regarding landspreading emissions in parishes with large intensive livestock units, which would in reality export some of their excess manure (see also Chapter 10)

It is useful to consider the spatially distributed analyses of the variance or coefficient of variation of the emissions, calculated at the 5 km grid level from the 1 km model results. These show the local variability due to environmental conditions and average farming practice, i.e. where high variability (within a 5 km gridsquare) is hidden through aggregation from the processing level (1 km) to the publication level (5 km resolution). In order to quantify the locational accuracy of emission sources in the model output within each parish, access to spatially distributed holdings level data would be required. This approach would also be beneficial for addressing cross-boundary issues between parishes.

- *Temporal uncertainties in emission estimates*

  Regarding annual emission estimates, as provided in this thesis, there are two main uncertainties to consider:

  a) Annual average emissions hide inter-annual variations, mainly due to in environmental conditions, i.e. drier/wetter and/or warmer/colder conditions.

  b) Variations of emission source strength also occur at a seasonal, monthly or daily level, due to meteorology as well as agricultural activities. These activities, e.g. manure spreading or livestock grazing, are linked to the natural growing cycle as well as to a diurnal cycle.

  Further development of inventories on shorter time scales is needed to address questions of atmospheric NH_3 budgets in atmospheric transport models.

Chapter 10
Modelling the variability of emission source strength: possibilities for further work

10.1. INTRODUCTION

It has been shown in the previous chapters that several elements of the spatial variability of NH_3 emissions have not been treated in the national scale model. This applies to both the spatial distribution of NH_3 sources and the spatial variability of NH_3 source strength estimates. The expected variability of NH_3 emissions per source unit ('emission factors') has been described in Chapter 2, and linked with environmental factors and the influences of differing farming practices. Farming practice varies on a local scale between different farmers and in relation to individual conditions on each farm, and also on a more regional scale between different parts of the UK. It has been pointed out in Chapter 9 that local scale differences cannot be resolved in a model for the whole UK, due to lack of detail in the information available. This is due to a number of reasons, from legal issues such as confidentiality and disclosivity, to data collection and processing issues. In addition, for the 5 km grid resolution required by the national inventory, individual deviations from the average conditions within a grid square are likely to be smoothed out, except for very large farms (see Chapters 8 and 9).

This chapter considers ways to estimate systematic regional trends in emission per source unit and explores how these trends may be incorporated into the basic model developed for this study. Such an approach should not only improve the model regarding the spatial distribution of NH_3 emissions, but also have an influence on the (not spatially resolved) NH_3 emission totals for specific regions of the UK. It is also likely that the total UK emissions would be changed by moving away from applying average emission strength per source unit. A way to achieve this is through the construction of matrices for spatially variable factors influencing NH_3 source strength, which are then applied in the emissions model.

This move away from average emission source strength estimates, as outlined above, can only be achieved in a spatially distributed model, by taking into account variations in conditions relevant to emission source strength for each gridsquare in the country. In the following sections, examples of this approach are considered, where some of the variables responsible for deviations from the mean emission source strength estimates have been identified. Thus, the following factors may be incorporated into an individual estimation of emission source strength for the conditions at each location:

- The influence of environmental factors (such as regional variations in temperature and its influence on grass growth, the housing duration of grazing livestock and the solubility equilibrium);
- The influence of agricultural practice (such as regional variations in fertiliser application rates or landspreading of livestock manures from intensive non-land based enterprises);

There is, however, more work required before the model can be revised to include these effects.

10.2. THE REGIONAL VARIABILITY OF CATTLE AMMONIA EMISSION SOURCE STRENGTH

Ammonia emissions from agricultural livestock vary depending on the N excretion per animal as well as the environmental conditions and agricultural practice under which the animals are kept and the manures are managed (see Chapters 2 and 3). The distinction between grazing and housing periods is particularly relevant in this respect, since the latter provide much larger NH_3 emissions per unit time (see Chapters 2 and 3). In the UK, this aspect is mostly relevant for cattle, which are grazing outdoors for part of the year and housed for the rest of the year. Other livestock are mostly either housed all year (pigs and poultry) or outdoors all year (sheep).

The emission source strength estimates used in this thesis, as well as those of other recent studies (see Chapter 3), assume that cattle in the UK spend on average half the year outside grazing and half the year housed (Section 3.2.1.). Assuming that daily excretion rates do not vary between housed and grazing animals, cattle thus produce 50% of their total excreted N on the pasture and 50% in housed conditions. As discussed in Sections 2.2. and 3.2.1., daily emissions during the grazing season are much lower than during the housing season (13.6% of the total annual emissions for 0.5 years; TFEI, 1996), due to higher volatilisation rates under housing conditions (see also Section 2.2.). In addition, the manure produced by housed cattle contributes to further NH_3 emissions through storage and landspreading losses. This effect would tend to increase emissions per animal in the colder northern and eastern parts of the UK, due to shorter grazing seasons than in southern Britain.

There are, however, other factors to be taken into account. Due to the colder temperatures in the areas with longer housing periods, the NH_3 emission rate is lower than in warmer southern and western areas with longer grazing seasons. This is estimated to offset the effect of the longer housing period to a certain degree.

The regional variability in cattle emission estimates is also dependent on the agricultural practice in the area. This variability in livestock husbandry is partially random, depending on preferences of the farmer and thus would need detailed data in order to resolve this. More systematic variability may be found in the intensity of the cattle enterprises, depending on the land capability and cattle type, i.e. dairy or beef. While the majority of dairy cows in the UK are kept more intensively and under similar conditions, there is more variety in beef production, ranging from extensive hill beef systems to intensive systems where the animals are mostly kept indoors. Thus the length of the grazing season is estimated to be more of an issue for dairy cows. The cows may be outdoors for a longer period in e.g. Cornwall than in Ayrshire and thus show decreased emissions due to the longer grazing season. This may however be offset against colder temperatures in Ayrshire, which will keep the overall emission rate lower. For beef cattle the situation is different: while it is colder in the north and east, the cattle may still spend proportionally more time outdoors than their southern counterparts. This can be attributed to the facts that a) the beef breeds used may be hardier and stay outdoors with supplementary fodder after the growing season (Chapter 2), and/or b) that the husbandry system may be less intensive and thus the lower N input may keep overall emissions down despite longer housing durations.

A way forward towards quantifying regional differences in NH_3 source strength estimates would be to develop a mechanistic model in terms of physical and chemical processes, which would help to explore the issues pointed out above in an integrated way. This would also include regional differences in N input through fertiliser application rates (see Section 10.3.).

In the following, the potential maximum length of the grazing season is isolated as a factor influencing the emission source strength for cattle, focusing mainly on dairy cows. The results must however be viewed in the context of the other issues outlined above.

In general, the maximum potential length of the grazing season may be used as a good indicator of the likely circumstances typical for different regions of the country. The maximum length of the grazing season on a farm is mostly influenced by environmental factors, in particular by climate (see Section 2.3.2.). Temperature is the most important variable for any attempt to determine the number of 'grass growing days', followed by precipitation (e.g. Gregory, 1964; Grigg, 1995). When the grass growth has started in spring, it takes an average of 5-6 weeks (Frame, 1992) before the grazing season can begin. With falling temperatures in autumn linked to lower levels of solar energy, grass growth decreases and cattle are moved off the pastures and housed for the winter. This is not only due to the lack of fodder, but also to avoid poaching and decreased health of the cattle in cold and wet weather. The latter is especially relevant for dairy cows, rather than the hardier beef cattle.

Thus, climate data provide an estimate of the maximum length of the grazing season, which can be taken as a suitable indicator for the minimum housing period necessary for dairy cows, rather than as an exact measure of actual housing period. This gives a lower limit to the emission source strength estimate that can be expected for dairy cows.

It has been shown that the grass growing season varies significantly between different regions of the UK (Section 2.3.2.), thus influencing the maximum length of the time the cows may spend outdoors grazing without supplementary fodder. If the annual grazing emissions and the emissions connected to housing (including storage and landspreading) are converted to daily emission source strength estimates and used as input into a simple model, the significance of the duration of the housing and grazing season, respectively, becomes apparent (Tables 10.1., 10.2.).

Table 10.1. Estimated daily emission source strength for grazing and housed cattle (after TFEI, 1996).

Emission source	Dairy cows ($g\ NH_3$-N day^{-1})	Other cattle ($g\ NH_3$-N day^{-1})
grazing emissions	17.8	8.9
other emissions (housing, storage &landspreading)	112.8	56.3

Table 10.2. Maximum potential length of the grazing season and its influence on annual NH_3 emission source strength estimates for cattle (emission source strength calculated with estimates from TFEI (1996), see Table 10.1.).

Max. grazing season (days)	Dairy cows (kg NH_3-N yr^{-1})	Other cattle (kg NH_3-N yr^{-1})
0	40.6	20.3
50	35.9	17.9
100	31.1	15.5
120	29.2	14.6
140	27.3	13.6
160	25.4	12.7
180	23.5	11.7
200	21.6	10.8
220	19.7	9.8
240	17.8	8.9
260	15.9	7.9
280	14.0	7.0
300	12.1	6.0
320	10.2	5.1
340	8.3	4.1
360	6.4	3.2

The total NH_3 emissions for a dairy cow rise by 0.95 kg N $year^{-1}$ for every 10 days the housing duration increases, using the source strength estimates provided by TFEI (1996). For other cattle, the annual emissions per animal increase by just under 0.5 kg N under the same conditions. Using the source strength estimates provided by the latest officially agreed inventory for the UK (BBSRC 1997b), the increases in emissions due to a shorter grazing season would be even larger. This is because according to their estimates the grazing emissions contribute even less to the total annual emissions per animal (10% for dairy cows, 12% for average cattle), while employing the same assumptions regarding the average duration of the grazing season.

In this context, it is relevant to consider the variation in the start of the grazing season between years. A cool spring with grass growth starting 10 days later than average for the whole UK would result in an increase of total cattle emissions of 6.9 kt (4%) compared with the average conditions assuming 183 days housing (estimates for 1996; after TFEI, 1996).

Spatially variable emission source strength estimates derived from the maximum length of the grazing season would not only have an effect on the total emissions in any given area. The variability of the grazing season would also have knock-on effects regarding the spatial distribution of the different subsources for cattle. This is because of the average management practices which the methodology tries to reflect in the source distribution model, which was designed to apportion the animals as NH_3 (sub)sources onto different landcover types (Chapter 5). Including spatially variable management practices into the model would require a) significant refinement of the model and b) the acquisition of relevant spatial data. For example, accumulated temperature maps for the UK as well as precipitation and altitude data would be required to model the maximum length of the grazing season. These data would also allow estimates to be made for inter-annual variability.

Summarising, it is emphasised that the contribution of the potential maximum length of the grazing season to the regional variability in emissions must not be seen in isolation, but modelled together with other factors causing variability. An example for this is the decrease of emission rates with lower temperatures, which is estimated to offset the higher emissions due to a shorter grazing season duration, and vice versa. Differences between beef and dairy cattle should also be taken into account. It is suggested that future work to resolve the regional variability in emission source strength estimates should be based on a mechanistic model of emissions. This model should take the physical and chemical processes contributing to the variability into account together with other spatially variable factors such as the N input into the model.

10.3. SPATIAL VARIATION IN N INPUT RATES AND CONSEQUENCES FOR THE UK AMMONIA EMISSIONS INVENTORY

Nitrogen input rates are the main influence on the magnitude of NH_3 emissions from crops and conserved grassland, whether they are applied as mineral N fertilisers or livestock manures. N application rates also have a considerable effect on emissions from livestock, through the N content in grass and fodder crops grown on the farm or concentrated feedingstuffs bought in by the farmers.

Variations in fertiliser N application rates are estimated to result in variations in emission source strength (see Chapter 2). This is not only an issue regarding average emission source strength estimates, but also influences the spatial distribution of emissions in the UK, due to large variations in agricultural practice. Any attempt at building this spatial variation of fertiliser N application rates into the spatially distributed emissions model will only be successful if systematic regional differences can be identified for Great Britain and modelled to a satisfactory degree.

The basic spatial NH_3 source distribution and emissions model used average values from the British Survey of Fertiliser Practice (BSFP, Burnhill *et al.*, 1997; see also Section 2.5.) to estimate emissions from crops and cut grass. The BSFP average N application rates are indirectly used in emission source strength estimates for grazing livestock (cattle and sheep, e.g. BBSRC, 1997b). The BSFP also analyses the total variability of fertiliser N application rates for England & Wales and Scotland (e.g. Table 2.10.), as well as in a more detailed format for different farm types such as arable farms, dairy farms, mixed cattle and sheep farms etc (Table 10.2.). However, Burnhill *et al.* (1997) do not provide spatially distributed estimates of fertiliser application rates.

For this study, the dataset used to derive the BSFP tables was made available by the Edinburgh University Data Library. The BSFP tables, as described in Chapter 2.5., are summaries derived from approximately 1500 farms, which are surveyed annually. These farms are selected from all the main holdings in GB through a stratified sampling approach. The main selection criteria employed are farm size and farm type, in order to reflect the variability in fertiliser practice across Britain (Burnhill *et* *al.*, 1996). However, some geographical stratification was achieved by ordering the holdings according to the 'parish-holding number' within each stratification cell before the random samples were taken from the cells.

Table 10.2. Fertiliser use on grassland for selected farm types in Great Britain in 1996 (after Burnhill *et al.*, 1997).

Category	England & Wales (kg N ha^{-1})	Scotland (kg N ha^{-1})
dairy farms		
grass < 5 years old	220	172
grass ≥ 5 years old	166	113
all grass	177	122
cattle & sheep farms		
grass < 5 years old	134	-
grass ≥ 5 years old	72	-
all grass	79	-
mixed farms		
grass < 5 years old	-	159
grass ≥ 5 years old	-	93
all grass	-	120
Farms in less favoured areas		
grass < 5 years old	-	115
grass > 5 years old	-	79
all grass	-	85

The detailed BSFP dataset provided an opportunity to analyse the spatial pattern of fertiliser N application rates from this large database, i.e. 1500 sampled farms every year, with a new random sample drawn every year. Work is ongoing at the Data Library to model and analyse the spatial patterns resulting from these surveys for all major crops. For the pilot study reported here, permanent grassland was chosen, as it is the most frequent agricultural cover type in Great Britain (e.g. Burnhill *et al.*, 1997). Thus it provided the most dense point data sample and hence the best basis for modelling and analysis. Permanent grassland was also chosen as it was expected to be one of the most likely cover types to show an interpretable spatial pattern, in relation to variations in farming practice due to environmental and economic factors. For this thesis it helps to explore the potential for modelling variability in the spatial distribution of livestock emissions with regard to fertiliser N input to the livestock farming system.

When attempting to create a continuous surface from irregularly spaced point data and thus to bring point and areal data into a common spatial framework for further modelling and analysis, a range of spatial interpolation techniques may be used (Burrough, 1986). The methodology selected for spatially modelling the N application rates in the first instance was inverse distance weighted interpolation (IDW) within a GIS environment (ARC/INFO), to create a continuous surface for Great Britain at a 5 km resolution. Other suitable interpolators would have been 'splines' or 'kriging' (e.g. Dubrule, 1984; Burrough, 1986), however IDW was chosen here as a simple and reliable approach for the pilot project (number of points = 12, max. radius = 200,000 m, power = 1). The success of a spatial interpolation technique depends to a large degree on the sample density and the representativeness of the sample points for the phenomenon under investigation. The sample density was sufficient for most of England and Wales and the lowland parts of Scotland. For the Highlands and Islands of Scotland, however, very few points were available, as was expected due to the sampling strategy employed (Figure 10.1.). The sample size drawn for any region is dependent on the density of farms within the region compared with the average farm density in Great Britain. Thus the sample for very extensively used agricultural areas, such as upland Scotland, is much smaller.

The resulting interpolated surfaces for 1995 and 1996 (Figures 10.2a and 10.2b.) show distinctive spatial patterns of fertiliser N application rates, which are similar between the two years analysed, at least for the areas with a good density of sample points. This indicates that there is a consistent spatial pattern in the data, which were derived from independent samples. However, for the 1995 surface very few data points were sampled for the far southeast of England, while the 1996 data showed a more even spatial distribution of sample points.

It should also be noted that in some areas the influence of single farms with contrasting practices, e.g. organic farms with no fertiliser N input in a high N area, is reflected in much lighter squares in dark surroundings in Figures 10.2a and b. In order to remove some of these random influences, the two datasets for 1995 and 1996 were combined. This leads to a more stable pattern by making single points less important and also by improving the sample density in spatially underrepresented areas such as the southeastern and northwestern parts of Britain. This approach was possible because of the similarity in the average N application rates in the BSFP tables as well as in the overall spatial patterns in the modelled surfaces, which suggests that there was very little real change in farming practice between the 2 years on average (Figure 10.2c).

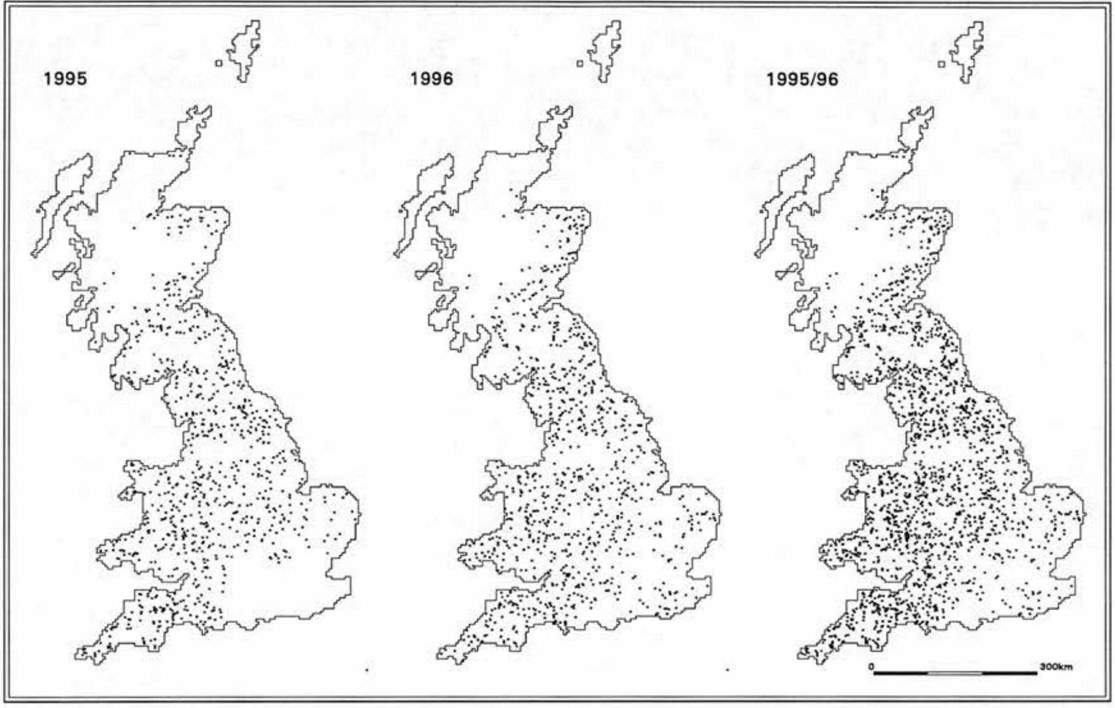


Figure 10.1. Spatial distribution of sample points for permanent grassland in the British Survey of Fertiliser Practice (BSFP; data source: Edinburgh University Data Library, pers. comm., 1996).

The combined surface for 1995/96 (as well as the surfaces for the individual years) shows a distinctive relationship between fertiliser N application rates and the environmental and farming practice conditions on the ground. For instance, areas with higher N application rates show a remarkable similarity to the pattern of intensively farmed areas with better land capability classes (compare with Figure 2.5.). Land capability is, in effect, a classification based on a combination of soil, topographic and climatic conditions, with the best land being most suitable for a wide variety of intensive agricultural uses.

However, further research is required to make use of these findings and to improve the NH_3 source distribution model, by including spatially variable N fertiliser input into the estimation of NH_3 source strength for cattle. The relationship between grazing emissions from cattle and sheep and N application rates have been studied in detail (e.g. Jarvis and Pain, 1990; Orr *et al.*, 1995; see also Chapter 3.2.1.). It cannot, however, be simply assumed that other livestock emission subsources (housing, storage and landspreading of manures) would increase at the same rate as grazing emissions with higher N input to the fodder. Furthermore, more detailed investigations of the BSFP data are needed to get better spatial information regarding the type of holdings distinguished in the sample design. This would hopefully enable a better identification and separation of spatial patterns in N application rates for dairy and beef cattle.

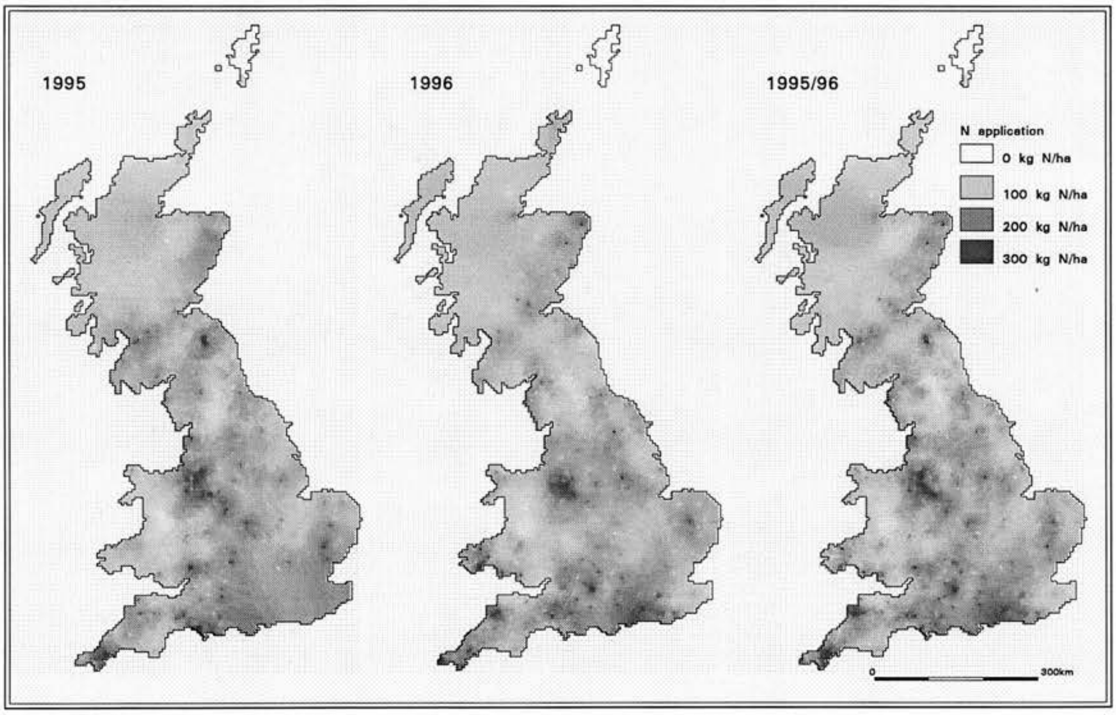


Figure 10.2 (a-c) Spatial variability of fertiliser N application rates (kg N ha^{-1} $year^{-1}$) for permanent grassland in Great Britain modelled from BSFP data for a) 1995, b) 1996, c) combined for 1995/96.

With the present surfaces (Figures 10.2a-c), the spatial pattern of fertiliser input rates shows that the main dairy areas (e.g. Cheshire and Staffordshire, Devon, Somerset, Dorset, Central Scotland) have larger N inputs than the beef areas (i.e. most upland and hill areas). Thus dairy farmers outside the main dairy areas may be underestimated regarding fertiliser N input to their fields in the BSFP sample, while N input by beef farmers in the main dairy areas may be overestimated with the present surfaces, if no further differentiation is made. Furthermore, not all the fields on a farm with mostly dairy cows may be used for feeding dairy cows, as some may be used for beef cattle or sheep instead. The data available for this pilot study provided average N inputs for all the fields under the same crop/grass category for each farm, and no information regarding the farm type at each data point.

In addition to permanent grassland, grass under 5 years old is also used for cattle grazing and hay and silage making. This would also have to be investigated and included in any attempt to provide spatially distributed fertiliser N application rates for use in an improved version of the spatial NH_3 emissions model. So far, it has been shown that there is a distinctive spatial pattern underlying the magnitude of N application rates for permanent grassland. This is expected to be of importance for reducing the uncertainty in spatially distributed NH_3 emission estimates for grazing livestock as well as for cut grass. Similar effects regarding the spatial variability of N application rates and thus NH_3 emission estimates can be assumed for arable crops.

10.4. VARIABILITY IN AGRICULTURAL PRACTICE REGARDING THE LANDSPREADING OF LIVESTOCK MANURES

In the NH_3 source distribution model developed in this thesis, the source sector distribution is scaled by the presence of animals, i.e. most of the emissions will occur close to where the animals are estimated to be. An exception to this is the landspreading of livestock manures from large intensive livestock units. Most of these farms produce too much manure to be able to spread it all on their own land. Many intensive pig and poultry farms have insufficient agricultural land which is suitable for landspreading, and therefore they export their manure outside the farm boundaries (see Section 5.3.). This excess manure may either be spread over agricultural land elsewhere, often 10s of kilometres away, or burnt in power stations (dried broiler manure only).

This issue needs to be taken into account when modelling the spatial distribution of NH_3 emissions using parish-based agricultural census data. The larger the farm size, compared with the land available for manure spreading on the farm, the more prevalent this problem is estimated to be.

The basic model described in this thesis distributes all NH_3 emitted from sources registered in each parish within the same parish's boundaries, and thus may overestimate local emissions in parishes with large intensive livestock units (see also Chapter 8). In order to improve the spatial distribution of NH_3 emissions further, it is necessary to consider modified approaches for the landspreading emissions for the parishes with very large numbers of pigs or poultry. A solution to this problem is to extend the spreading area outside the parish, linked to the availability of suitable agricultural land. This would be expected to smooth some peaks in the present spatial emissions inventory, which were generated by limiting the spatial distribution of NH_3 emissions to within the parish where the manures originated.

Several different issues have to be taken into account to implement such an approach. Firstly, with current statistics it is difficult to estimate the actual size of individual pig or poultry farms in parishes with large numbers of these livestock types. The disclosive holdings data per parish, which were made available for this study, could not be analysed for this purpose, due to agreements with MAFF and SOAEFD. At present, there may be parishes with a large number of small or medium sized pig farms, such as in Humberside, which all have individual strategies for dealing with their own manure. Conversely, a large number of the pigs or poultry registered in one parish might be entirely due to one farm, which is more likely to export some manure out of the parish. A revised approach on confidentiality and disclosivity issues would be necessary to make further progress here. For instance, suitable information would be provided by a register of all large installations of intensive livestock farming, as proposed by IPPC (EC, 1996),

Secondly, it is not clear at present what average ratio of animal numbers to farm size makes it likely that manure has to be exported for the purpose of landspreading, due to a lack of capacity on the farm's own land. The best possible approximate solution to this problem with currently available data sources is to set an estimated threshold for the size of a pig or poultry farm which is likely to produce manure in excess of the capacity they can put onto their own land. The landspreading emissions from these larger farms are then taken out of the basic parish-based emissions model and distributed separately onto nearby arable land and grassland.

Thirdly, it is difficult to anticipate where the landspreading of manures from these large farms is taking place, and how far away the 'sphere of influence' of such a farm would reach, or if the manure is being burned. For the latter approach, information on the spatial locations of any existing poultry power stations would also be useful. There are several possible solutions for the issue of finding suitable locations for spreading the surplus manure.

a) A simple approach would be a spatial decay function, with the farm itself in the centre of the manure spreading area, and distributing the manure in a concentric manner, with lower application rates away from the centre, until all manure is spread.

b) A more realistic solution is to spread the manure, again starting at the farm, but spreading only onto suitable land which has not had been 'filled' to capacity with manure applications from other farms, such as cattle enterprises or smaller pig, poultry or mixed farms. This iterative process is repeated in widening circles until all the surplus manure has been spread.

The main advantage of this second approach is that it simulates reality better, because it takes average agricultural practice on other farms into account. Thus it would distribute the surplus manure exported from the large intensive livestock units only onto land which has not reached its manure 'carrying capacity' by the prior allocation of manure from other farms in the area. This approach would require a system to map the spreading of all manure from different livestock types onto each 1 km square, and converting the manure produced by different livestock types to maximum recommended spreading rates for appropriate landcover types.

Another issue that needs to be resolved is the location of the proposed centre squares within each parish where the landspreading submodel starts. It is difficult to model the spatial location of pig and poultry units within each parish in the first place, as livestock at farm types is not landbased. The real location(s) of the farm(s) larger than the parish-based manure-to-land threshold, which would activate the landspreading submodel, are not identifiable as such in the emissions map. Instead, emissions from these farms would be spread to all the grid squares within each parish, where the farm(s) are most likely found. In order to approximate a single representative location for each parish, where the spreading of all surplus pig and poultry manure from the parish can be centred, it is suggested here to use the centre of gravity of all possible locations within the parish. Thus, before the land-spreading submodel is activated, the excess manure from all pig/poultry farms above the threshold size would be accumulated as one 'source' at the centre of gravity of the parish.

Further work is required to identify the best possible way to implement the solutions suggested above. It is expected that the inclusion of this or a similar submodel into the basic parish-based NH_3 source distribution and emissions model would have a significant impact for selected areas of the country. Improvements in the emission estimates due to this methodological development would primarily be in the areas with the highest NH_3 emission peaks as well as with the highest variability.

10.5. DISCUSSION AND CONCLUSIONS

Clearly the present model for the national NH_3 emissions inventory is simplistic in many respects, especially regarding the emission source strength estimates ('emission factors') that were applied, assuming average conditions to be valid over the whole country. In this chapter, ways of improving the model by introducing spatially varying emission source strength estimates have been investigated, dependent on environmental factors and farming practice in the UK.

A key objective for further work is to quantify the regional variability in emission source strength which can be expected in the UK due to the issues discussed above. This allows an assessment of the impact of this variability on the magnitude as well as the spatial distribution of NH_3 emissions in the UK.

The case studies described in this chapter are representative for the type of improvements that may be made to the model, given more time to research the underlying processes and their links with NH_3 source strength, as well as access to relevant datasets. Thus this chapter has focused on outlining avenues for further work rather than solving the problems identified.

Examples are the influence of fertiliser application rates on the magnitude of emissions from crops and conserved grassland as well as grazed pastures, or the effect of the length of the grazing/housing periods on NH_3 emissions from livestock that spend part of the year outdoors. The latter is especially relevant for cattle in the UK. Care should be taken, however, to take differences in farming practice such as beef production and dairying in different regions into account, rather than relying on environmental factors only for the definition of the grazing/housing period.

Both fertiliser application rates and the maximum potential length of the grazing season can be modelled, provided access to the relevant spatially distributed datasets (BSFP, climate/weather data) and further research into process-based links of these factors with NH_3 source strength. Their effects on the spatial variability of NH_3 emissions should however not be modelled in isolation, but in a mechanistic model of emission source strength in terms of physical and chemical processes. This would for instance include the effect of temperature on the emission rate from livestock manures etc., which may offset the effect of the grazing season duration.

A further issue in modelling NH_3 distributions linked to farming practice is that large intensive pig and poultry farms may export manure from the parish of origin, due to insufficient land available for spreading. Currently, the emissions from these sources are redistributed entirely within the parishes in which the farms are registered for agricultural census purposes. This leads to a potential overestimation of emissions in the parish of origin, as manure from large intensive livestock farms with limited land for manure spreading may be exported to other farms outside the parish where the manure was produced. Alternatively, in some instances broiler manure is removed from the farms to be burned in power stations. It is suggested to develop a sub-model for redistributing emissions from the landspreading of manures from large intensive livestock farms over a wider area. In order to ensure realistic results, an investigation of farming practice regarding e.g. the distances manure is transported, the ratio of farm area and number of animals is proposed. It would also be useful in this respect to have access to IPPC farm data rather than having to identify thresholds where the proposed submodel would improve the spatial inventory.

Chapter 11
Discussion and conclusions

11.1. INTRODUCTION

The main focus of this study has been to improve the spatial distribution of NH_3 (NH_3) emission estimates and explore the uncertainties involved in the resulting inventory for the UK. This has been achieved through the development of a new methodology at the national scale (5 km grid resolution). An implicit secondary objective was the expected improvement in the results of atmospheric transport and deposition models, which use the new inventory as their key input dataset. The use of the new methodology was shown to have a large effect on the spatial distribution of the emission sources and thus on the reliability of the inventory, compared with previous inventories.

Particular attention has been paid to uncertainties in the input data as well as in the model assumptions, and these are discussed in detail and quantified where possible. Uncertainties of the national model regarding average input data, especially concerning agricultural practice, were explored further through a comparison with a fine scale model for an approximately 5 km by 5 km study area. This local inventory was developed from detailed data on livestock husbandry and arable farming at a field scale.

The following sections summarise the work undertaken for this thesis and discuss the main objectives, methodology issues, results and uncertainties of the models developed at both the national and local scale. Furthermore, the application of the spatial inventory as an essential component in the development of atmospheric transport models is discussed, and the derived maps of predicted atmospheric concentration are compared with measurements. This was achieved by comparing the model results with independent measurements from the National Ammonia Measuring Network (Sutton *et al.*, 1998c). Avenues for further work are considered and prioritised, followed by the key conclusions of this thesis.

11.2. DEVELOPMENT OF A NATIONAL SPATIALLY DISAGGREGATED AMMONIA EMISSIONS INVENTORY

11.2.1. Objectives and background

One of the main objectives of this thesis was the development of a new spatially distributed UK NH_3 emission inventory, for both agricultural and non-agricultural sources. Given that NH_3 plays a major role in the eutrophication of N sensitive ecosystems and the acidification of soils and water bodies, it is crucial for target-oriented abatement to have reliable quantitative information on the location of emission sources. This spatial perspective is especially important, as previous studies have shown large variations in NH_3 emissions and deposition over the country (e.g. Kruse *et al.*, 1989; Eager, 1992; Sutton *et al.*, 1995).

Given that spatial emission inventories generally use average source strength estimates, it was considered essential to review the basic chemical and physical processes involved in NH_3 emissions, as well as to identify environmental factors and elements of agricultural practice which influence emission source strength. Factors contributing to the variability in source strength include climate, topography, soil conditions, fertiliser use on crops and grassland, and aspects of livestock husbandry, such as grassland management and grazing issues, housing, manure storage and spreading (see Chapter 2). Emphasis was put on the importance of these issues when estimating the emission source strength, and when attempting to quantify the potential for spatial variability in source strength over the UK, dependent on local/regional conditions.

A comparison of source strength estimates by different studies (e.g. Asman, 1992b; ECETOC, 1994; DoE, 1995; Sutton *et al.*, 1995; TFEI, 1996; BBSRC, 1997a and b) showed that recent figures appear to agree more closely than was the case with earlier studies (Chapter 3). This has been found for both individual source strength estimates (e.g. per animal), and for the total magnitude of UK NH_3 emissions (Chapter 3). However, despite an increased convergence of source strength estimates, large uncertainties remain, and several key issues have yet to be resolved. These include widely differing estimates of N excretion by agricultural livestock, especially for sheep and pigs, and to some extent for cattle. The amount of N excreted by animals is one of the main variables for estimating source strength from livestock, as it provides the basis for estimating N losses from manures in a process-based model, which takes the flow of N through the manure management system into account (e.g. Cowell, 1998). Furthermore there is a lack of data in some parts of the N flow model, which makes accurate estimates difficult. The final choice of a set of source strength estimates for the inventory (DoE, 1995) was determined by the need to bring the results in line with commitments to UK governmental requirements and commitments to international organisations such as EMEP/ CORINAIR. For these reasons the source strength estimates of DoE (1995) were selected for the main model scenarios, and amended with more detailed information on the source sector components of TFEI (1996).

11.2.2. A new methodology for spatially disaggregating ammonia emissions over the UK

Previous spatially resolved NH_3 emission inventories were based on existing general distributions of the main sources, i.e. livestock numbers and crop areas from the Agricultural Census, at a 5 or 10 km grid resolution. It has been shown here that the use of these pre-prepared spatial datasets (Hotson, 1988) employed in previous studies causes a systematic shift of emissions from the main agricultural areas to hill and upland areas within each census unit (i.e. civil parishes). The resulting overestimate of emissions in extensively farmed areas, especially in upland and hill areas leads to larger atmospheric concentrations and deposition estimates when the inventory is applied in atmospheric transport models. Conversely, emissions in intensively farmed areas are underestimated in these approaches. It is emphasised here how important a realistic spatial distribution of emission sources and sink areas is with regard to NH_3. This is because NH_3 is highly reactive, and a large proportion of the emissions is deposited close to the sources (e.g. Sutton *et al.*. 1998b; Pitcairn *et al.*, 1998). Thus the value of estimated critical loads exceedance maps for abatement decisions etc. diminishes substantially if these issues are not taken into account.

The new model developed here divides livestock emissions into component sources for grazing, housing and manure storage and landspreading of manures (Chapters 4, 5). These sub-sources are then weighted and spatially redistributed onto suitable land cover types within each parish, rather than equally spread over all agriculturally used land. A sub-model for grazing emissions employs a weighted approach using relative stocking densities of grazing animals, and thus distributes grazing emissions according to the quality of grazing land available.

The new model involves multiple vertical and horizontal integration of large spatial datasets. In order to facilitate fast and efficient data processing and thus allow the efficient production of different scenarios, but also to perform in-depth analyses of the resulting spatial inventory, the model was implemented in a 'hybrid' approach, linking GIS and a purpose-built FORTRAN77 model (Chapter 4). This choice of implementation environment allowed the manipulation of the spatial input data as well as the results within the GIS, while benefitting from the rapid data processing capabilities of the linked FORTRAN model. Similar approaches have been employed in many environmental models (e.g. Nyerges, 1992; Reyes *et al.*, 1993), although the work here represents the first application of this approach for modelling NH_3 emissions.

11.2.3. Results of the new ammonia emission model for the UK

The results show that the new methodology removes previously estimated apparent emissions from extensive grassland, especially hill and upland pastures as well as other semi-natural areas, and concentrates them more realistically in intensive agriculturally used areas (Chapters 6, 7). The significant improvement in the quality of the new modelled emission estimates has been confirmed through a recent validation study with a large number of measurements (see Section 11.3.2.). Further improvements in the match between the modelled and measured data can be attributed to the updating of the agricultural statistics used in the model from 1988 to 1996.

The availability of 1996 census data also provided an opportunity to achieve a better spatial resolution of data for a substantial area in England and Wales. This was possible because more detailed disclosive data were made available for this project, under the condition that the model output would be non-disclosive. The difference between using disclosive and non-disclosive data in the model is substantial, especially in areas with relatively few farms per parish (due to small parish sizes and/or large farm sizes). This was shown for a study area in eastern England (Section 9.2.1.). In addition to these improvements, an NH_3 emission inventory for Northern Ireland was compiled and spatially distributed for the first time, as were other miscellaneous (non-agricultural) sources (Chapters 5, 6). Thus a complete and up-to date NH_3 emissions inventory for the whole UK has been produced.

The results of the new UK inventory were analysed in detail regarding total magnitude and the spatial distribution of contributions by different source sectors over the country. Furthermore, the dominant source(s) for each 5 km grid square were identified (Chapter 6). This revealed distinctive patterns in the magnitude as well as the spatial aspects of NH_3 emissions from the different source sectors. Overall NH_3 emissions in the UK are dominated by cattle, regarding the total magnitude as well as spatially. Areas dominated by cattle as well as areas with no distinctive dominant source are generally characterised by emissions in the medium range (5-30 kg N ha^{-1} in a 5 km grid square). The areas where sheep or crop emissions provide the largest contribution to the total are generally typified by low total NH_3 emissions of 1-5 kg N ha^{-1} in a 5 km grid square. Pig and poultry dominated areas and some urban areas typically show estimated emissions at the higher end of the range of total emissions per 5 km grid square (10-120 kg N ha^{-1}).

The results of these and similar analyses are strongly dependent on the accuracy of the input data for the underlying emissions model, especially source strength estimates and redistribution rules. A revision of the model with e.g. larger emissions from sheep as suggested in Section 3.2.2. or a more sophisticated spatial redistribution of non-agricultural sources would shift the balance between source sectors. The general spatial pattern revealed by these analyses, however, appears to be stable, and highlights the importance of pigs and poultry as well as some non-agricultural sources as the sectors that are most likely to cause locally acute adverse effects to the environment. This can be explained by the very high animal density on large pig and poultry farms, which is not linked to land-based stocking densities as is the case for grazing livestock such as cattle and sheep, and thus causes very high emissions per unit area.

11.2.4. Temporal Changes in ammonia emissions 1969-1996

A further objective of the work undertaken here was to study changes in the magnitude and the spatial pattern of UK NH_3 emissions over time. Inventories were developed and analysed for the periods of 1969-1988 and 1988-1996. A substantial increase in the use of fertiliser on crops and grassland between 1969 and 1988 points towards a large increase in the total NH_3 emissions for this period, despite relatively stable livestock numbers (with the exception of a large increase in the number of sheep). The higher emissions can be attributed mainly to increased NH_3 emissions from livestock due to an increased use of fertilisers on grassland and fodder crops. This has been shown by, for example, Jarvis and Pain (1990), Jarvis and Bussink (1990) and Orr *et al.* (1995).

It is, however, difficult to derive reliable source strength estimates for past/ different husbandry conditions and N levels without a more process-based approach to following the N flow through the manure management system. This issue needs to be addressed further to provide more certain estimates. An increase in the number of emission 'hot spots' on the 1988 map also indicates a shift towards larger, more intensive enterprises, especially in the pig and poultry sector.

For the period of 1988-1996 the overall magnitude of livestock emissions appears to have remained more or less unchanged, with a slight decrease of fertiliser use and thus crop emissions. In the spatial context, substantial relative and absolute changes have been shown, both due to relocation or new developments of large pig and poultry farms. Large relative changes have also occurred in low emission areas, where they may nevertheless be of considerable importance regarding deposition and impacts on nitrogen-poor ecosystems in the vicinity of sources.

Scenarios of potential future changes due to the implementation of abatement measures were also developed, focusing on large intensive installations as proposed by the EC Directive on Integrated Pollution Prevention and Control (IPPC; EC, 1996). By concentrating on the pig and poultry sectors, which provide the highest local emission peaks, the benefits of abatement centre on the worst affected areas. Although not a deliberate part of the planning of IPPC, this finding suggests that the focus of IPPC on intensive pig and poultry farming would be expected to have proportionally more environmental benefits than equivalent (kt year^{-1}) control of NH_3 emissions from cattle, sheep or crops.

11.2.5. Uncertainties in the UK ammonia emissions inventory

Particular attention was paid to issues of uncertainty in the work undertaken. The need for the evaluation and assessment of uncertainties is driven by the requirement to estimate the reliability of the model output, which is used in atmospheric transport models and may also be used for decision support regarding abatement measures.

Dealing with the issue of uncertainty in NH_3 emission inventories has so far been limited to uncertainty in the source strength estimates by previous authors (e.g. Kruse *et al.*, 1989; Asman *et al.*, 1992b; Sutton *et al.*, 1995). In this study, the investigation of uncertainties in the spatial inventory has been taken further. The main sources of uncertainty in the spatial model input data and the assumptions behind the redistribution model were identified and quantified where possible (Chapters 9, 10). Uncertainties in the source strength estimates per unit source were not only evaluated as such (Chapter 3), but also regarding the sensitivity of the spatial pattern and magnitude of the model output to changes in source strength (Section 9.3.). The main causes of uncertainty in the UK NH_3 emissions inventory were found to be:

- the smoothing effects due to the aggregation of farm census data to parishes (Modifiable Areal Unit Problem, MAUP),
- the variability in source strength over the UK in relation to environmental factors and agricultural practice compared with the average source strength estimates used in the model,
- the general uncertainty in the set of average source strength data used in the model (as discussed in Chapter 3),

- the average rules in the model for the whole country regarding the spatial distribution of emissions, and
- inter-annual and intra-annual variability in the emission source strength dependent on environmental conditions and agricultural practice.

Avenues for further work towards quantifying and resolving these main uncertainties have been suggested and discussed in detail (Chapter 10; see also Section 11.5.).

11.3. SCALES OF SPATIAL VARIABILITY OF AMMONIA EMISSIONS

A suitable spatial resolution is crucial for a realistic spatial distribution of NH_3 sources and sinks as well as for modelling atmospheric transport, deposition and critical loads exceedance. This is especially important as, in contrast to other pollutants, a large proportion of the NH_3 emitted is deposited in the immediate neighbourhood of the source rather than transported over long distances. Sutton *et al.* (1998b) estimated that the average fraction of NH_3 recaptured by dry deposition within the same 5 km gridsquare in the national inventory was between 8-50%, depending on landcover types across Great Britain.

The present spatial resolution of the UK inventory of 5 km provides a marked improvement compared with the older 10 and 20 km inventories for the UK (e.g. Kruse, 1986; Kruse et al., 1989), or the 50 km EMEP inventories (e.g. Berge *et al.*, 1995), which hide the substantial spatial variability within each gridsquare. The most suitable scale and spatial resolution for any model output are determined by the model input and the assumptions and calculations inside the model as well as by the purpose of the model, due to the uncertainties in both the data and the model. On the other hand, data should be used at the best possible resolution available to avoid introducing further uncertainties into the end product. It is therefore argued here to distinguish between a 'processing level', which allows model calculations to be performed at the greatest level of detail, and a 'publication level' at a coarser resolution. The subsequent aggregation of the 'processing level' output to the 'publication level' provides more robust results, prevents users from taking the more uncertain fine resolution data at face value, and also facilitates meeting the present disclosivity requirements stipulated by MAFF and SOAEFD.

The model discussed here was therefore developed at a 1 km resolution to match the resolution of the input data available ('processing level'), and the results were aggregated to the 5 km level ('publication level'). The 1 km results are at present potentially disclosive regarding the agricultural census redistribution and therefore excluded from publication. With further treatment to ensure confidentiality they could however be a valuable source of additional information, as they provide a statistical representation of the likely spatial distribution at the sub-5 km level. An emissions inventory at a 1 km resolution would show more detail in the agricultural landscape than the 5 km grid results, such as narrow intensively farmed valleys surrounded by upland and hill areas (compare Section 5.3.).

The variance and the coefficient of variation within each 5 km square were calculated from the 1 km results to quantify the differences between the 2 scales (Section 9.4; Figures 9.5., 9.6.). The variance map highlights the gridsquares with a large range in emissions, which occur mostly near intensive pig and poultry farms. The % coefficient of variation is also high in the same areas (>150%), and additionally in upland areas where intensively farmed land is spatially mixed with extensively used upland and hill areas, such as in the Scottish Highlands. Low values (< 20 %) are found in some grassland dominated cattle farming areas (Figure 9.6.). This leads both to areas where emissions are underestimated (intensive agricultural land) and areas where emissions are overestimated (hill areas, semi-natural areas) within the 5 km squares concerned. Such features argue for the continued development of methods to improve the spatial resolution of NH_3 emission estimates.

A change in the publication level from the present 5 km squares to a 1 km model would bring about a 25 fold increase in data volume, which would provide an opportunity for more detailed analyses, model testing and validation, which is not possible under the present agreements. This substantial increase in the data volume may be too detailed for present atmospheric transport models at the national scale, but would be highly valuable for application to specific sensitive areas or for comparison with site-based monitoring data (see Section 11.3.2.).

At a sub-1 km scale (e.g. field scale, Chapter 8), the magnitude of NH_3 emissions is highly variable in space, and the range between maximum and minimum values in any 1 km square may be even larger than the range of the 1 km square estimates within a 5 km square. This has been shown for a local study area containing a large poultry farm in central England (5 km by 5 km), where emissions were estimated to range from 0-8,000 kg N ha^{-1} $year^{-1}$ in 1996 (Figure 8.1.). The study undertaken here provides the first field level NH_3 emissions map for the UK. A point source study has been published at a fine resolution in the Netherlands (Boermans and Erisman, 1993), however not at a field level.

The extreme spatial interplay between sources and sinks in the landscape substantially affects atmospheric concentrations and deposition at the local level (Figures 8.5., 11.4.). This is further discussed in Section 11.4.1.

Studies linking the national inventory with local scale inventories are of great value for the validation of the model input data as well as the assumptions built into the national scale model. This is especially important regarding agricultural practice (see also Section 11.4.3.).

11.4. APPLICATION AND VERIFICATION OF THE MODELLED AMMONIA EMISSION ESTIMATES

11.4.1. Application of the derived ammonia emission inventories as inputs to atmospheric transport and effects assessment models

One of the main reasons for developing spatially distributed NH_3 emission inventories is that they are an essential input dataset for atmospheric transport models and hence maps of N deposition and impacts derived from these models. The results of this thesis were used as input to such models in the ADEPT project (Ammonia Deposition and Effects ProjecT; Sutton *et al.*, 1997), at both the national and the local scale.

At the national scale, the mapped NH_3 emission estimates provided input to the FRAME (Fine Resolution AMmonia Exchange) model (Singles, 1996; Singles *et al.*, 1998; Sutton *et al.*, 1998b). This atmospheric transport model was developed as a multi-layer trajectory statistical model at a 5 km grid resolution, thus incorporating essential detail in the horizontal as well as the vertical gradients of NH_3. This permits detailed modelling of the behaviour of NH_3, regarding atmospheric concentrations, dry and wet deposition. Figure 11.1. shows the predicted ground level atmospheric concentrations of NH_3 for 1988, derived from the emission maps presented Chapters 6 and 7 (Figure 6.6.).

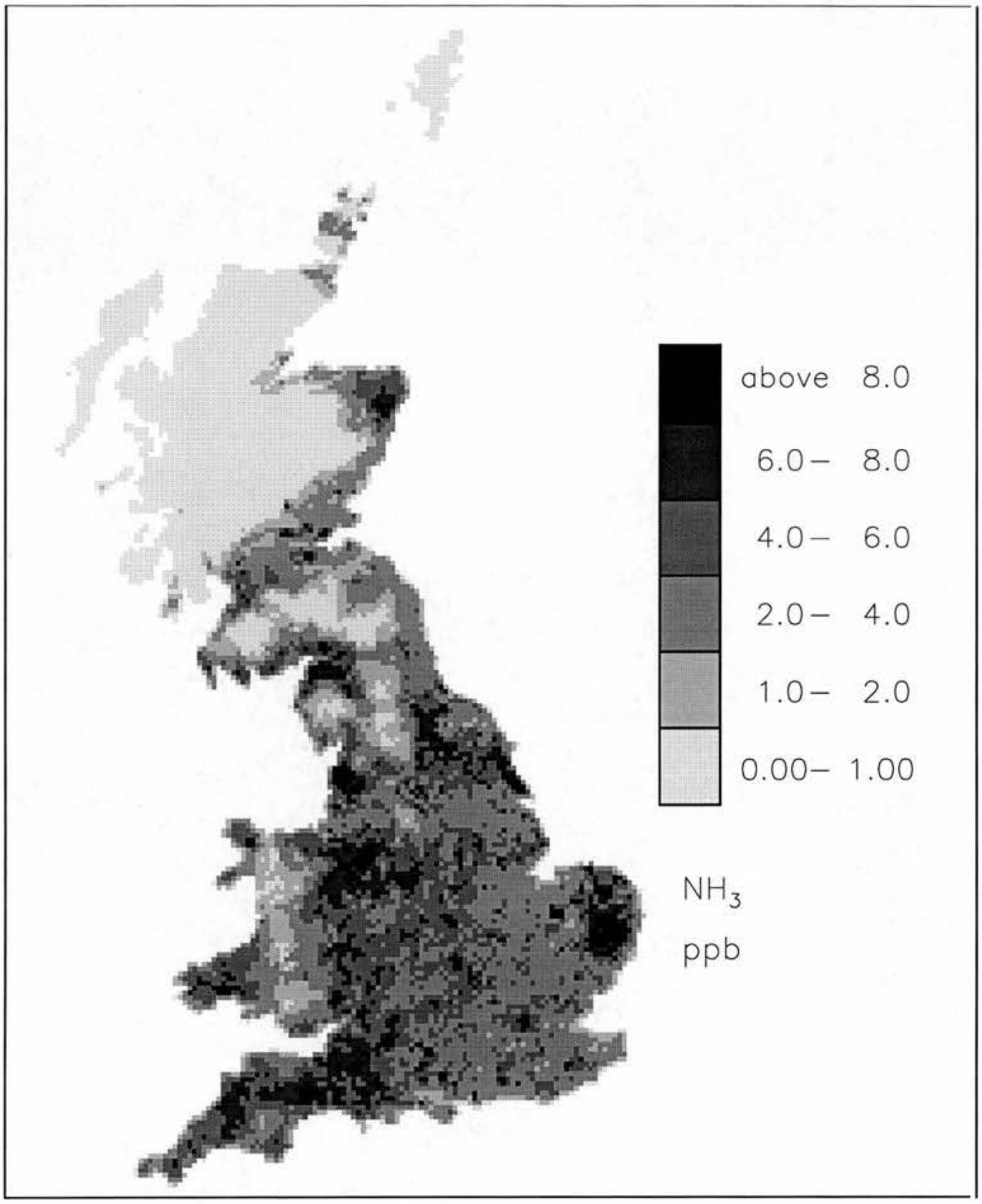


Figure 11.1. Modelled ground level atmospheric concentrations of NH_3 for 1988 (from Singles *et al.*, 1998).

FRAME also incorporates a canopy resistance model for dry deposition in relation to the landcover types on the ground. Hence the model not only provides average deposition estimates for each square, but also separate estimates for different landcover types. This is exemplified in Figure 11.2., which shows estimated dry deposition of NH_3 to forest ecosystems in Great Britain. Maps such as Figure 11.2.

can be further used to assess the impacts of NH_3 deposition to certain types of natural and semi-natural habitats/vegetation communities using the critical loads approach. Based on the results of FRAME together with estimates of wet deposition of NH_3 and deposition of NO_y to account for the total N deposition, further spatially distributed results were derived. The total N deposition maps were then compared with critical loads maps, thus producing patterns of critical loads exceedance in Great Britain as a whole (e.g. Figure 11.3.).

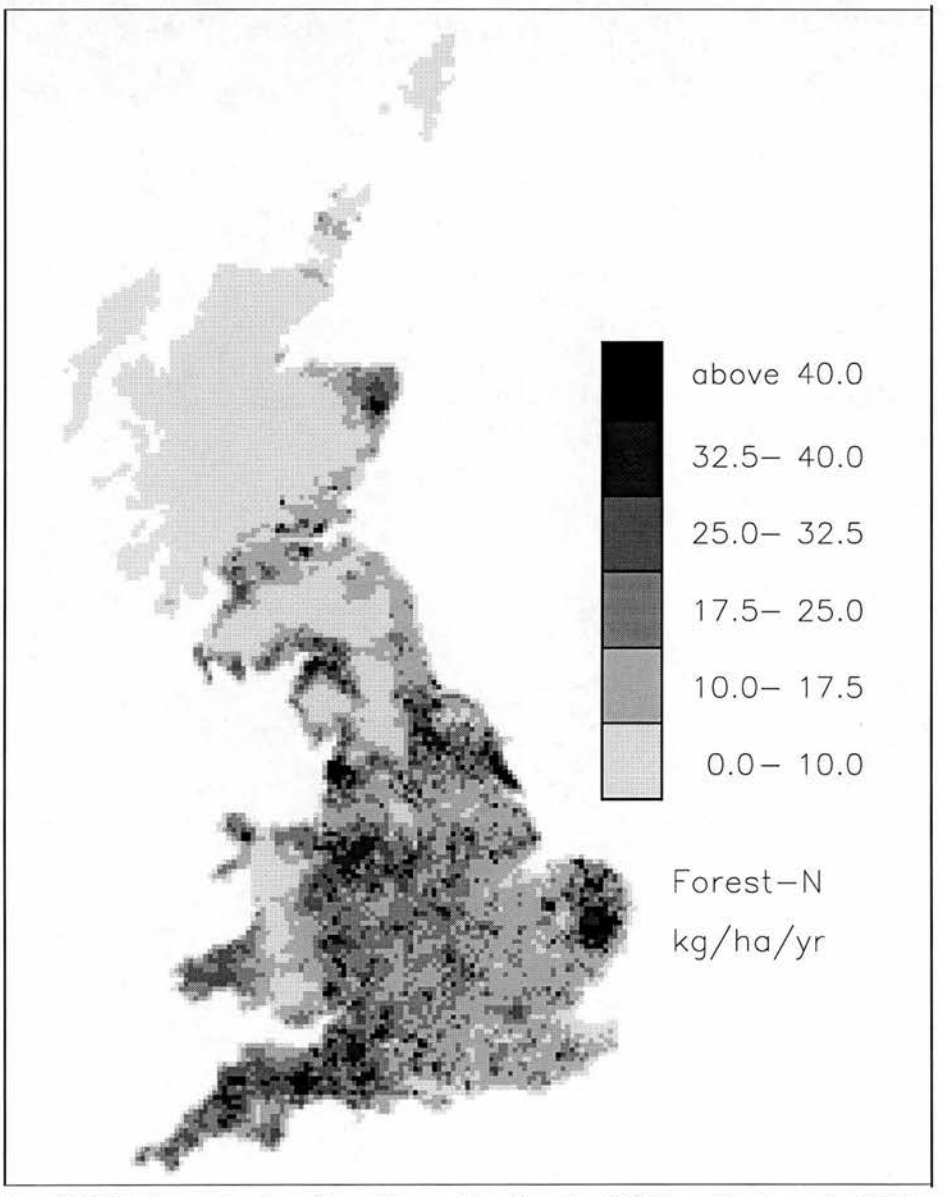


Figure 11.2. Estimated dry deposition of ammonia to forest in 1988 (from Singles *et al.*, 1998).

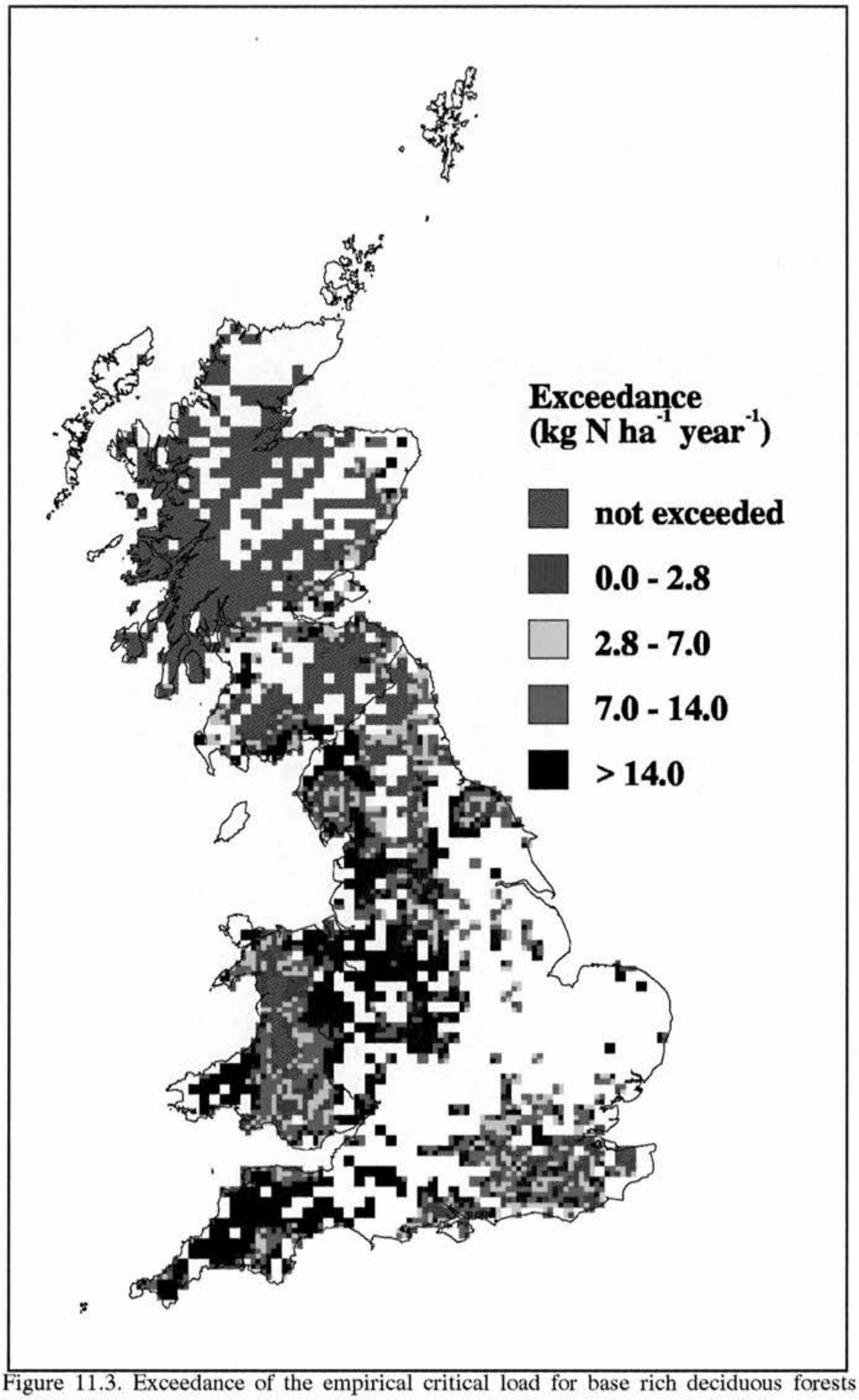


Figure 11.3. Exceedance of the empirical critical load for base rich deciduous forests from NH_x deposition (1988-1992), without any abatement measures included); from Sutton *et al.*, 1998b.

Further work is ongoing, with the FRAME model results being updated with the 1996 emission inventory. This also includes Northern Ireland for the first time (see Section 6.7.).

On the local scale, the emission inventory described and discussed in Chapter 8 was employed in the LADD (Local Area Dispersion and Deposition) model (Hill, 1998, Sutton *et al.*, 1998b; see Section 8.5.). The fine resolution of this model and the use of detailed landcover data (to provide dry deposition velocities) enabled a closer investigation of the variability within a national 5 km grid square. The high local variability in emissions and thus in air concentrations (0.1-85 μg m^{-3} for 1993; see Figure 8.5.) is mirrored in a high variability in deposition (Figure 11.4.). Local deposition of ammonia emitted from sources in the study area is estimated to be largest close to large point sources such as livestock housing and manure storage facilities, as well as in semi-natural areas such as the edges of woodland.

Figure 11.4. Estimated deposition rates (μg NH_3 m^{-2} s^{-1}) from the LADD model for the local scale study area (Sutton *et al.*, 1997).

11.4.2. Comparison of the FRAME model estimates of NH_3 concentrations with the National Ammonia Monitoring Network

Although initially tested against a limited number of NH_3 air concentration measurements (RGAR, 1997), the UK NH_3 emissions inventory and the derived FRAME NH_3 concentration estimates needed a more comprehensive validation. This is being achieved through a new monitoring network, which provides spatial patterns of measured air concentrations (Sutton *et al.*, 1998). The National Ammonia Monitoring Network was set up in 1996, and the results of the 1988 emission inventory (see Chapter 6) were used together with other considerations to select representative sampling locations (Sutton *et al.*, 1998c). At present, this UK wide network contains 72 sites (see Figure 11.5.) and has been providing monthly NH_3 air concentration measurements for more than 2 years. For validation purposes, the network NH_3 concentration data were compared with the modelled air concentrations as estimated by FRAME.

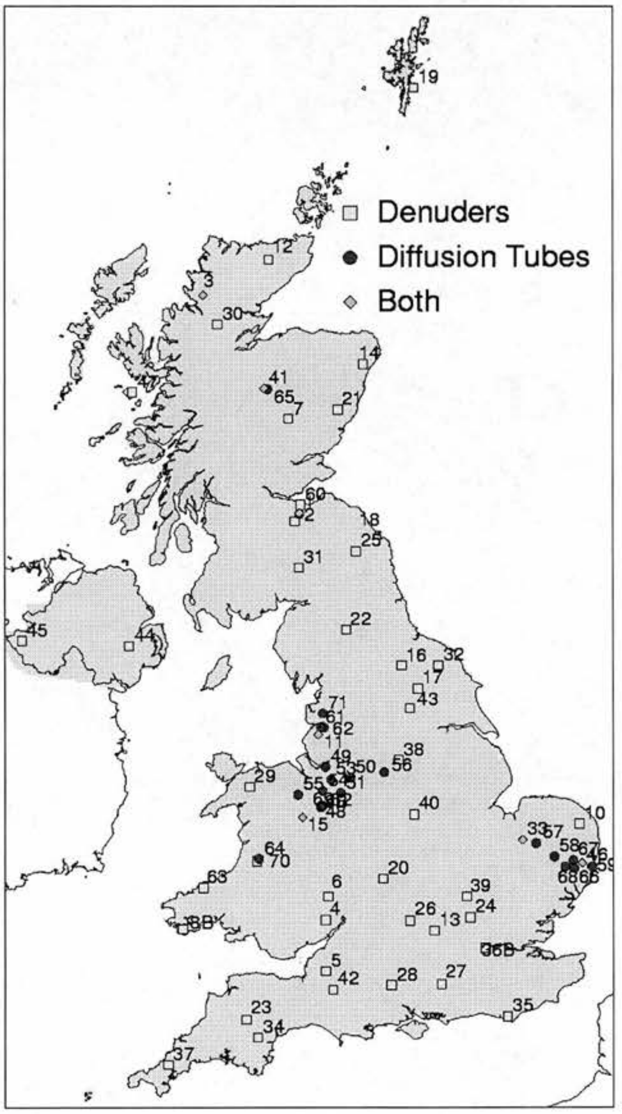


Figure 11.5. Map of the NH_3 network sites over the UK (from Sutton *et al.*, 1998c).

The measurements were compared with the FRAME air concentration fields for 1988 (modelled using emissions data derived with the old methodology) and 1996 (using emissions data modelled with the new methodology). It was anticipated that the 1996 model would agree more closely with the measurements, since a) the more recent parish source statistics match with the monitoring period, b) the new emission model was expected to provide a better spatial location of the emission sources.

An overlay of the network concentrations onto the FRAME predictions for both years shows close correspondence, regarding both magnitude and spatial variability on a national scale (Figure 11.6.). Overall, the FRAME concentrations and the network data agree that the upland sites generally have the smallest measured concentrations, while the largest concentrations occur in pig and poultry dominated areas.

Sutton *et al.* (1998c) showed further that the new inventory for 1996 (Figure 11.6b) provides improved agreement compared with the 1988 inventory (Figure 11.6a). This is largely due to the new spatial redistribution methodology developed, which locates sources in relation to the fraction of NH_3 emitted on different land cover types. Compared with the old methodology for 1988, this provided much smaller and more realistic emission estimates for hill ranges in agricultural regions. The clearest example of this is in the North Pennines (site 22; see also Figure 11.5.), where the measured concentration of 0.24 μg N m^{-3} agrees well with the model value of 0.30 μg N m^{-3} for 1996. In contrast, the FRAME predictions for the same site with the 1988 emissions data were much larger at 1.74 μg N m^{-3} and clearly overestimating the magnitude of emissions in this area.

Summarising the results of Sutton *et al.* (1998c), changes in the modelled concentrations between 1988 and 1996 against the measurements for each site show improved model estimates for 1996 for the pig and poultry areas (e.g. sites 33, 66, 67, 68), as well as for upland and hill sites (sites 22, 25, 55, 64; Sutton *et al.*, 1998c). This is an important validation for the new methodology, and shows that, as a consequence, much smaller effects of NH_3 dry deposition would be expected in such hill areas than predicted using the 1988 emission inventory.

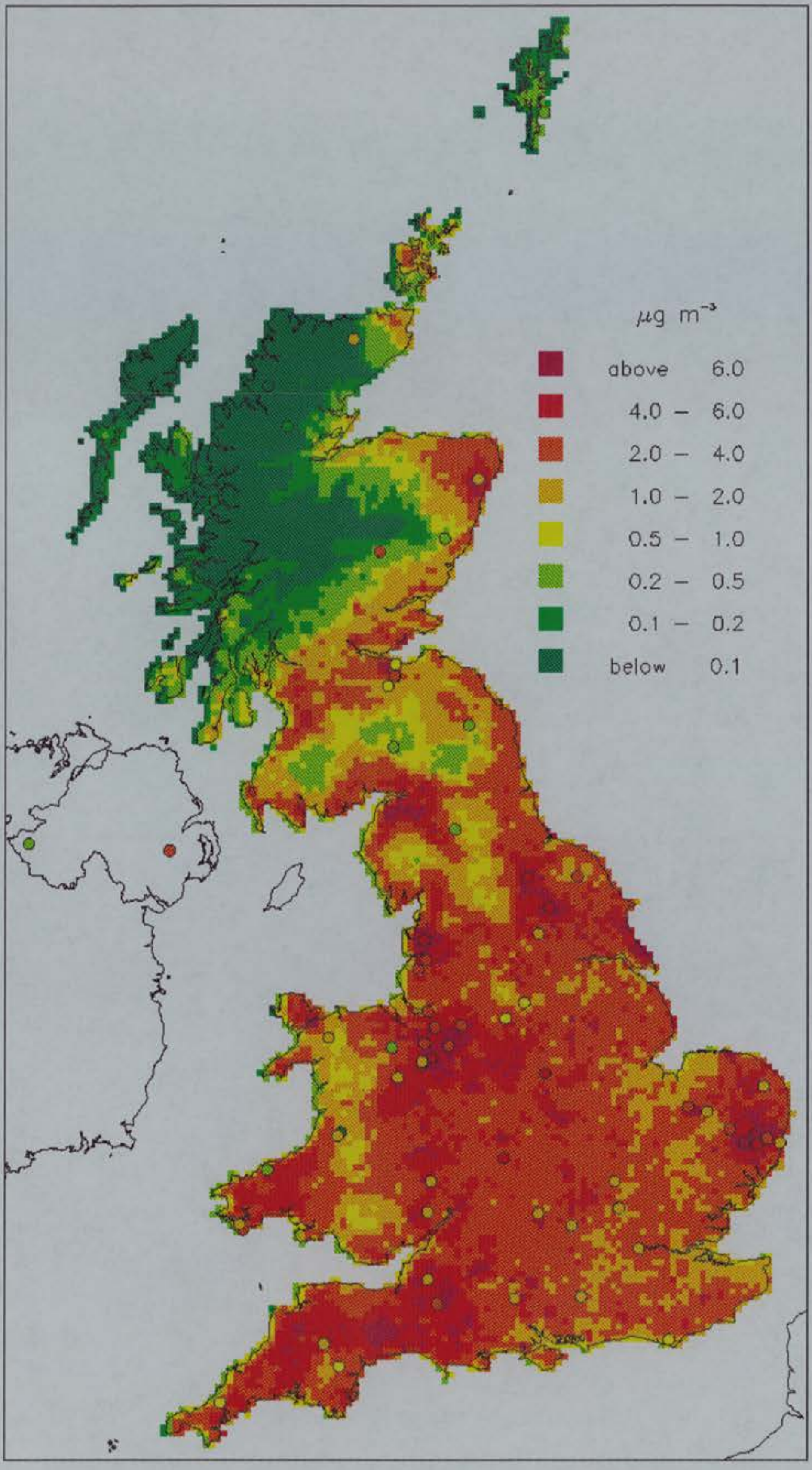


Figure 11.6a Comparison of monitoring network results (points) with FRAME model estimates using the 1988 emission inventory developed with the old methodology (from: Sutton *et al.*, 1998c).

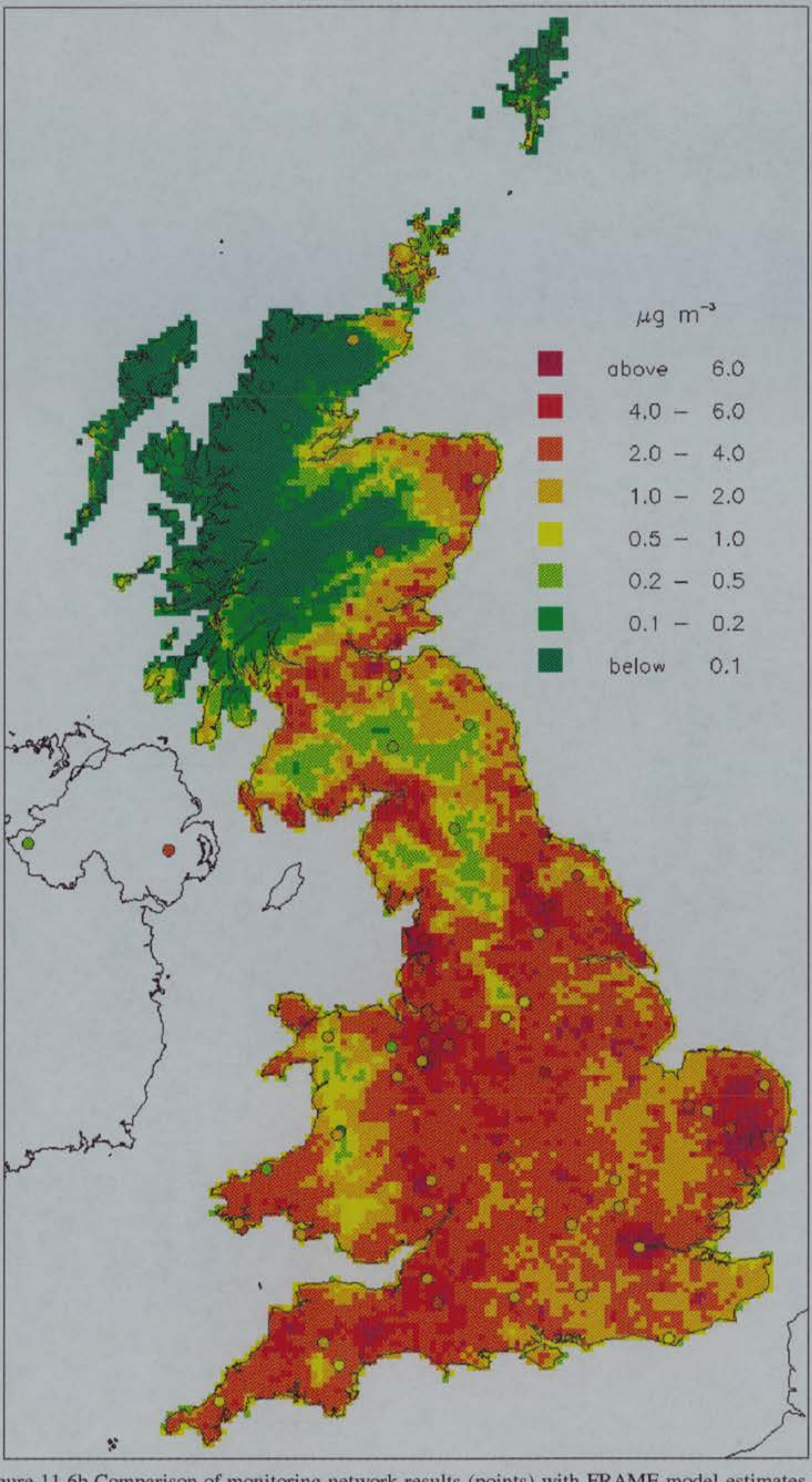


Figure 11.6b Comparison of monitoring network results (points) with FRAME model estimates using the 1996 emission inventory developed with the new methodology (from: Sutton *et al.*, 1998c).

In contrast, the worse model estimates for urban areas in 1996 are also visible in Figure 11.5. The measured concentrations in London (site 36), Sheffield (site 38) and Edinburgh (site 60) are much smaller than the FRAME estimates for both years. This most likely reflects the concentration of non-agricultural emissions in areas with high population concentrations, due to the simple methodology used for the spatial distribution of non-agricultural sources (i.e. largely scaled by population density). This indicates the need to refine the methodology used in the spatial distribution model of these miscellaneous sources.

Differences between FRAME and the network data are also evident in some source areas where the model appears to overestimate, and in some remote areas, where the model seems to underestimate concentrations. These differences may be explained by a combination of factors related to local issues (scatter, distance of monitoring site to local sources, reliability of 5 km emission estimates), meso-scale issues (variation in bias of estimated emission factors for different source sectors cattle, sheep, pigs, poultry, crops, non-agricultural sources), and national scale issues (compensation point, model diffusion scheme).

Meso-scale spatial variability in NH_3 concentrations was investigated by comparing a transect across East Anglia, where emissions are dominated by pigs and poultry, with FRAME estimates using the 1996 NH_3 emissions data. Figure 11.7. shows close agreement between the two independent datasets for this study area, both in absolute terms and spatially.

There is however still significant variability at the meso- and local-scales that has not been fully resolved. The existence of sub 5 km variability has also been demonstrated by emission estimates at the 1 km level as shown in Figures 5.3 and 5.4., as well as at the field level (Chapter 8). The 1 km maps (Section 5.3.) distinguish very clearly between hills and valleys, and this helps to explain some apparent anomalies in the comparison of the monitored data and the 5 km model estimates. An example for this are the relatively high measured concentrations at Glenshee (Site 7), which are most likely biased by the relative location of the site in the corresponding 5 km grid square, i.e. in a valley surrounded by extensively grazed hills. This aspect of local variability is not only important regarding emissions and air concentrations, but also air concentrations, but also when effects of N deposition on semi-natural ecosystems and critical loads exceedance are considered. Thus care should be taken when generalised 5 km model estimates are compared with point measurement data, especially in areas with high local variability in source strength.

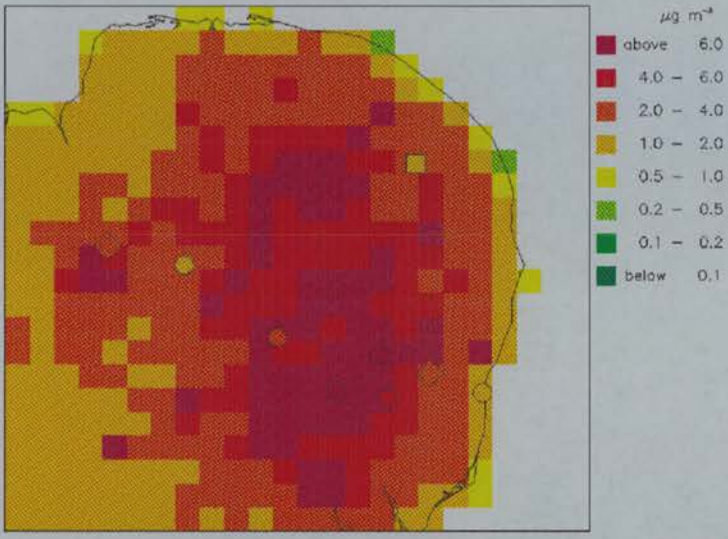


Figure 11.7. Map of East Anglia showing (as coloured points) the NH_3 concentrations determined by the monitoring network on a background of the FRAME 96 estimates developed with the new emissions methodology. Round symbols indicated parallel sampling by DELTA system (Denuder for Long Term Ammonia) and diffusion tubes, diamonds by diffusion tubes only, and the square by DELTA only (from Sutton *et al.*, 1998c).

It should be noted that the measured concentrations at similar sites with a high proportion of sheep emissions are also larger than the model estimates. This may again be due to the location of the monitoring sites in relation to the spatial patterns of sources, as discussed above, but may also indicate an underestimation of the NH_3 emission rates from sheep in the model.

Sutton *et al.* (1998c) recommend local-scale studies for both upland and lowland areas to examine site representativity within the 5 km grid squares of FRAME. They suggest setting up local networks for selected 5 km grid squares, where the local variability is likely to explain the difference between the network data and the FRAME estimates. This would provide a better understanding of both site representativity and the reasons for differences between the measurements and atmospheric transport models.

Despite these constraints at the local level, comparisons between the network data and FRAME provide a sound validation of the model estimates, particularly for the new 1996 results. The overall magnitude as well as the spatial distribution of NH_3 air new 1996 results. The overall magnitude as well as the spatial distribution of NH_3 air concentrations provided by the model are reproduced in both agricultural and remote areas.

11.4.3. Comparison of the national and local emission inventories

The fine scale emission model (Chapter 8) with detailed data on agricultural practice for a 5 km by 5 km study area was developed for 3 main purposes: a) to study the variability within a sample national 5 km gridsquare, b) to provide input data for a fine scale transport and deposition model (Sections 8.5. and 11.3.1.), and c) to assist in validating the average assumptions that were built into the national scale model. The study area is kept anonymous due to disclosivity issues involved.

It has been shown that the 5 km results are robust compared with the aggregated emissions from the local study area, in fact the 2 inventories match remarkably well. A closer look at the underlying models and the input data at both scales revealed the following:

- Individual farm estimates of livestock emissions differed widely at the local level, especially regarding animal housing duration and fertiliser application to pastures, however these discrepancies were evened out over the 5 km gridsquare.
- For the aggregated total of emissions from fertiliser application to crops and conserved grassland, good agreement was found again between the two inventories. The figures matched less closely for individual crops, with some receiving consistently higher or lower N rates in the study area than the national average as estimated by the British Survey of Fertiliser Practice (Burnhill *et al.*, 1997).

These results suggest that the national inventory may underestimate emissions in intensive agricultural areas, where the average source strength may be too low in the UK model, especially where large pig and poultry enterprises are present. This may be caused by a) higher than average source strength per animal connected with intensive farming practices and b) by smoothing out the emissions from localised large sources over whole parishes. Conversely, emissions estimated with average national source strength figures for extensive areas may be too large, thus providing overestimates for these areas. Examples of possible overestimation are suckler beef production and hill sheep farming, compared with e.g. intensive dairying and lowland sheep farming. It is recommended to use further local scale studies in areas with varying agricultural source sectors and intensity to assist in the further development and validation of the UK national emission inventory, and to evaluate the impacts of abatement measures at a local scale.

11.5. KEY AREAS FOR FURTHER WORK

Key areas of further work have been identified during the course of the project undertaken here. They include:

1. *A strong need for a spatial process-based model of emission source strength.* This should include the effects of environmental factors and different agricultural practices over the UK and facilitate the validation of emission source strength estimates, as well as assist in quantifying the range of uncertainty due to the influence of these factors. The results of such a study could be used further to derive a set of spatially variable emission source strength data, which could be incorporated into the source distribution and emissions model. Improvements in the spatial estimates are expected from including e.g. spatial climate data, which would have implications on the housing duration for some livestock categories, and spatially distributed N input to crops and grassland (see Chapter 10). Using a process-based model rather than treating these factors individually is especially recommended as a way forward here, as the different factors interact and partly offset each other. Such an approach would also aid in the derivation of improved estimates of emissions for inventories of past years (e.g. to derive more certain source strength data for 1969), as well as provide a tool for predicting future changes due to abatement measures under different conditions (or for different parts of the UK).

2. *Validation and improvement of the national UK inventory.* This can be achieved through the acquisition of data and the development of local scale emission inventories for several sample areas with varying NH_3 source sectors and

agricultural intensity. Such an approach would also be useful for modelling and evaluation of the impacts of abatement measures at a local scale.

3. *Estimates of the intra-annual and inter-annual variability of NH_3 emissions.* It has been shown that large uncertainties regarding the temporal resolution exist in the present annually averaged UK inventories. These are especially relevant when the modelled emissions are used in atmospheric transport and deposition models. This is due to the high temporal variability of NH_3 emissions, which can be linked to their origin in mainly agricultural processes with highly seasonal trends, as well as to the variability in environmental conditions affecting emission from sources present in the landscape. The process-based model could again provide a basis for this, especially for resolving issues of inter-annual variability.

4. *Development of sub-models allowing manure removal from the parish of origin to be quantified.* The present spatial redistribution and emission model has been designed to work within the spatial units of the Agricultural Census counts, i.e. civil parishes of variable size and shape. Distributing NH_3 emissions outside the originating parish would provide a means of accounting for landspreading emissions from large intensive pig and poultry enterprises, which is believed to occur frequently. Improved versions of the model should also reflect the fact that broiler manures from some areas are incinerated in power stations, and thus entirely excluded from landspreading.

11.6. CONCLUSIONS

Development of a spatially distributed ammonia emissions inventory for the UK

1. The distribution of NH_3 air concentrations and deposition over the UK is characterised by a high spatial variability. Previous studies have highlighted that this is due to a) the fact that NH_3 is highly reactive with a large proportion deposited close to the sources, and b) the high spatial variability in the distribution of NH_3 sources over the country.

2. Reliable quantitative information on the spatial pattern of NH_3 emissions is especially important with regard to atmospheric transport models, as spatially

distributed emission inventories provide the primary input data source to these models.

3. A new model has been developed in this thesis to spatially distribute NH_3 sources more realistically than earlier studies, using agricultural census data, average N fertiliser application rates to crops and grassland, landcover data and NH_3 source strength estimates. In contrast to previous methodologies, the new approach employs a spatial model specifically tailored to NH_3, rather than a more general allocation of agricultural sectors (livestock categories, crops etc.). These two approaches differ since the spatial probability distribution of NH_3 emissions is different from the spatial probability distribution of the sources.

4. The new model takes into account the different spatial distribution patterns from agricultural component emission sources, such as livestock grazing, livestock housing and manure storage, landspreading of manures and fertiliser application to crops and grassland. These component sources are then weighted by the magnitude of their emission source strength and distributed onto suitable landcover types at a 1 km resolution over Great Britain. A sub-model for livestock grazing emissions employs a weighted approach using relative stocking densities, thus distributing grazing emissions according to the quality of pastures available.

5. It has been shown that the best results are achieved by developing the model at the spatial resolution of the model input data (1 km grid), and subsequently aggregating the output to a resolution suitable for publication (5 km grid). This distinction between a "processing level" and a "publication level" ensures that, on the one hand, calculations are performed at the greatest level of detail, to avoid the introduction of further uncertainties into the results. On the other hand, the more robust 5 km output level helps in ensuring the present disclosivity standards stipulated by MAFF and SOAEFD and precludes the mis-use of the 1 km results as definitive rather than statistical estimates.

6. Due to the need for multiple vertical and horizontal integration of large spatial datasets, the most suitable implementation environment was chosen to link a

Geographical Information System (GIS) with a purpose-built FORTRAN77 model. This combined the complementing advantages of both approaches, i.e. the capacity of the GIS for the spatial manipulation of model input data as well as in-depth analysis of model output, and the fast and efficient data processing capabilities of the FORTRAN77 model to compute multiple scenarios of NH_3 source redistribution.

7. The results of the new UK NH_3 inventory were analysed in detail regarding the total magnitude of emissions, the spatial distribution of contributions by different source sectors such as cattle, pigs and poultry etc. In particular the identification of dominant source sectors for each gridsquare revealed distinctive patterns:

   - Overall NH_3 emissions in the UK are dominated by cattle. Areas dominated by cattle and areas with no distinctive dominant source generally show emissions in a medium range of 5-30 kg N ha^{-1} for 5 km grid averages.

   - Areas where sheep or crop emissions provide the largest contribution to the total are generally characterised by low total NH_3 emissions of 1-5 kg N ha^{-1} for 5 km grid averages.

   - Pig and poultry dominated areas and some urban areas typically show estimated emissions at the higher end of the total range (10-120 kg N ha^{-1} for 5 km grid averages). This can be explained by the high animal density on intensive farms, which is not linked to land-based stocking densities as is the case for grazing livestock.

8. A comparison of the resulting new inventory with previous models shows that emission estimates have been decreased to more realistic levels in extensively used areas such as upland and hill pastures, and concentrated in intensively used agricultural areas. This has major implications for the estimation of critical loads exceedances for intensive versus upland areas. In upland areas, exceedances will be less than previously estimated, whereas in semi-natural ecosystems in intensive agricultural areas they will be larger than previously estimated.

9. New sets of agricultural census data were made available for 1996 for the whole UK, thus allowing the development of a spatially distributed NH_3 emissions

inventory for Northern Ireland for the first time. Combining the updated agricultural emissions inventory with new spatially distributed estimates of non-agricultural NH_3 sources modelled in this thesis resulted in the most comprehensive UK NH_3 emissions inventory to date.

10. Both the overall magnitude and the spatial distribution of NH_3 emissions presented in this thesis are strongly supported by a comparison of air concentration fields derived from the new model with the results of the National Ammonia Monitoring Network. This has been shown both at the national scale and in a regional study in East Anglia.

11. Ammonia emission scenarios for the past (1969, 1988), present (1996) and future (spatial implications of potential abatement measures) were calculated, to study changes in the magnitude and spatial pattern of UK emissions over time. It has been shown that emissions increased substantially from 1969-1988, due to intensification of UK agriculture. This can be linked to increased fertiliser input to crops and grassland, with the largest contribution a consequence of livestock being fed a higher N diet. Between 1988-1996 changes in the overall magnitude of UK emissions were relatively small, however some significant localised changes in the spatial distribution were found. These spatial differences can be attributed to changes in the spatial distribution of agricultural activities, as well as to the improved spatial resolution of the 1996 agricultural census data.

Development of a local ammonia emissions inventory at the field scale

12. A field scale emissions inventory was developed with detailed data on agricultural practice for a 5 km by 5 km study area in central England, to provide estimates of local variability in emissions and to validate the average assumptions built into the national scale model.

13. A comparison of the national (5 km grid) and the local (field scale) inventory has shown that the model results are generally robust. However, a closer look at the underlying models and input data at both scales showed that much local variability is hidden in the 5 km model results. Differences in agricultural

practice on livestock farms (e.g. fertiliser application rates to pastures, livestock housing duration) and consequently different source strength estimates were however evened out over the 5 km sample square. On arable farms, fertiliser application rates for individual crop types vary considerably within the study area, but good agreement was found on average.

14. Other important findings at the local scale include a very high spatial variability in NH_3 emissions. This is critical for the assessment of impacts of NH_3 deposition from local sources, which were estimated from the inventory via a local atmospheric transport model. It has been shown that the largest deposition rates occur in the vicinity of local sources and over semi-natural areas in close proximity to sources, such as forest edges.

Uncertainties in spatially distributed ammonia emission inventories

15. It has been argued that identifying uncertainties in the presented NH_3 inventories, both at the national and at the local scale, as well as estimating the magnitude of these uncertainties is an essential part of the modelling process. In previous studies, only uncertainties in the applied NH_3 source strength estimates were considered. In this thesis the main sources of spatial uncertainties were also evaluated, and quantified where possible.

16. The main causes of uncertainty in the national inventory were found to be:

    - smoothing effects in the agricultural census data due to the spatial aggregation to parishes (MAUP),
    - variability in source strength over the UK with regard to environmental factors and agricultural practice,
    - the average rules regarding the spatial distribution of NH_3 sources employed in the model, and
    - the inter-annual and intra-annual variability in source strength depending on environmental conditions and agricultural practice.

17. A quantitative assessment of the modelled spatial uncertainty of the 5 km NH_3 emission estimates was carried out by calculating the % coefficient of variation of from the underlying 1 km. This showed high values of >150% in areas with intensive pig and poultry farming, as well as at the boundary between intensively farmed lowland areas and extensive upland and hill areas. Low values around 20% are typical for some grassland areas with predominantly cattle farming.

18. It is suggested here that the national NH_3 inventory may underestimate emissions in intensive agricultural areas, e.g. intensive dairying areas, where the average source strength estimates used in the national model may be too low. In extensively farmed areas with low N input, such as suckler beef or hill sheep farming, the national average source strength estimates are likely to result in overestimates of NH_3 emissions.

19. Underestimates in NH_3 emissions for some areas may be caused by the smoothing of emissions from localised large sources such as pig and poultry farms over whole parishes. This leads to overestimates in emissions in the remaining area of the concerned parishes.

20. The uncertainties identified above have been suggested as a basis for further work. These include:

    - the development of a spatially distributed process-based model for NH_3 emissions which takes the variability of environmental factors and agricultural practice over the UK into account;
    - the development of sub-models for manure movement between parishes, which is mainly relevant for intensive pig and poultry farming.
    - a closer investigation of the temporal dimension in the model to resolve intra-annual changes in emissions, as well as inter-annual changes, the latter in conjunction with the process-based model.
    - further validation of the national model through detailed local studies at the field scale.

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Appendix A

Sample Agricultural Census forms

RESTRICTED - COMMERCIAL

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD

Agricultural and Horticultural Census: Return for 3 June 1996

Notice requiring information to be completed and sent back by 10 June 1996

I

Agricultural Census Branch,
Rm 124, Foss House, Kings Pool,
1 - 2 Peasholme Green, York. YO1 2PX
Telephone: York (01904) 455284 or 641000

Under the Agricultural Statistics Act 1979 (as amended by the Agriculture (Amendment) Act 1984), the Minister of Agriculture, Fisheries and Food requires you to complete this form in respect of the land you occupy. **This is a legal requirement.** Under Section 4 of the 1979 Act, penalties may be imposed on any person who knowingly or recklessly gives false information, or who without reasonable excuse fails to provide information. No information you give on the form can be published or otherwise disclosed without your prior written consent, except as specified in section 3 of the 1979 Act.

The valuable information you supply will be used extensively by the Ministry, the industry and the European Union.

- Please read the Notes for Guidance carefully before completing this form.
- **Some changes have been made to the land questions this year arising from the need to gain information on the effect of the Agricultural Tenancies Act 1995.**
- The information you give should relate to the position at 3 June 1996 except where otherwise stated on the form.
- Please return the form by 10 June 1996 in the pre-paid envelope.

Notes for Guidance are provided. The enclosed yellow copy is for you to keep.

S J Holding, Statistician

In correspondence please quote your holding number

FOR OFFICIAL USE ONLY

B.R.	163	
I.C.	164	
L.S.	165	
M	167	
I.M.	168	

↑

166 Total recorded area shown above in hectares (1 hectare = 2.471 acres)

PLEASE CHECK THESE IMPORTANT POINTS FIRST

TOTAL AREA If the total area of your holding **is not as printed** on the address label above please give the correct area in box 169 opposite and account for the difference on page 6. *Exclude* land on which the keep is let to you on a seasonal basis - see enclosed NOTES FOR GUIDANCE.

	hectares
169	•

ADDRESS AND POSTCODE If any part of the address label above is incorrect or if there is no postcode please enter any corrections in the appropriate boxes on page 6.

RESULTS Provisional results will be published in a Statistics Notice at the end of August 1996 and final results will be available in December 1996. Enquiries to: Lynne Thom, Room 133B, Foss House, Kings Pool, Peasholme Green, York Y01 2PX on York (01904) 455332

RETENTION COPY Enclosed is a yellow copy of this form for your retention. Census forms are kept for a limited period. Requests for copies cannot normally be met; you may find it useful to complete and hold your retention copy.

HELP **If you need any help with completion of this form, please write to the address at the top of the page, (or telephone our Help Desk on York (01904) 455284 or main switchboard on York (01904) 641000) quoting your holding number.**

AFTER COMPLETING THE FORM PLEASE SIGN THE DECLARATION AT THE FOOT OF PAGE 6

AREA OF HOLDING AND MAIN LAND USES (enter all areas to the nearest 0.1 hectare) JUNE 1996

- see enclosed NOTES FOR GUIDANCE
- growers of horticultural/protected crops may find it useful to complete pages 4 and 5 first

CROPS AND FALLOW

			Item	hectares
Cereals for combining	Wheat		11	•
	Barley	Winter Barley	12	•
		Spring Barley	13	•
	Oats		14	•
	Mixed corn		15	•
	Rye		16	•
	Triticale		33	•
Maize			17	•
Potatoes - all crops *including* seed			19	•
Sugar beet not for stockfeeding			20	•
Hops			21	•
Horticultural crops - *exclude* mushrooms (to agree with item 249 on page 4)			22	•
Field beans			23	•
Peas for harvesting dry - human consumption or stockfeed			27	•
Other crops for stock-feeding	Turnips and swedes		24	•
	Fodder beet and mangolds		25	•
	Kale, cabbage, savoy, kohl rabi and rape		26	•
	Other crops (not grass) used for stock-feeding. Enter total area in box 28 and specify each crop and its area below:-		28	•
Rape grown for oilseed - *exclude* oilseed rape grown on land set-aside under an official payment scheme		winter	29	•
		spring	36	•
Linseed - *exclude* linseed grown on land set-aside under an official payment scheme			30	•
Flax - *include* only flax eligible for flax area aid (subsidy), *exclude* flax grown on Set-Aside Land			37	•
Other crops not for stockfeeding (see enclosed NOTES FOR GUIDANCE). Enter total area in box 31 and specify each crop and its area below:-			31	•
Bare Fallow - *exclude* Set-Aside Land			32	•
TOTAL Crops and bare fallow			35	•

GRASSLAND AND OTHER LAND

		Item	hectares
Grassland and rough grazing - *include* clover, sainfoin and lucerne	**Grassland** sown in **1992** or later	5	•
	All other grassland - *exclude* rough grazing	6	•
	Rough grazing on which you have **sole** grazing rights (see enclosed NOTES FOR GUIDANCE). Put grazed woodland in item 8.	7	•
Woodland - *include* grazed woodland on the holding, *exclude* woodland grown on land set aside under an official payment scheme		8	•
Set-Aside Schemes - Enter the total area of land Set-Aside under an official payment scheme, including any used to grow non - food crops.		34	•
All other land *excluded* above, e.g. paths, roads, yards, buildings, gardens, ponds, derelict land, recreational land		9	•

	Item	hectares
TOTAL Area of your holding (to agree with sum of items 35, 5 to 8, 34 and 9 above)	1	•

Questions on tenancy and ownership (to be completed by all) are now on page 5

GRASS GROWN FOR SEED

	Item	hectares
Area of grass already included in items 5 and 6 expected to be harvested for **seed** this year	40	•

IRRIGATION - *exclude* watercress

	Item	hectares
Total area of all outdoor crops on your holding which you are able to irrigate if necessary this year - *exclude* liquid manure spreading	43	•

(Questions on seasonal use of land are now on page 5)

PERSONS WORKING ON THE HOLDING

- *include* principal farmers and all other persons normally engaged on the holding at 3 June 1996
- *include* each person once only
- *include* persons engaged by you as trainees under an official scheme **only if** they are paid AWB rates or more otherwise see item 48 below
- see enclosed NOTES FOR GUIDANCE

FARMERS, GROWERS AND WORKERS

			Item	number
Principal farmer/grower or partner - if working on the holding		Whole-time	50	
		Part-time	51	
Wife or husband of principal farmer/grower or partner - if working on the holding			52	
Other partners and directors - if working on the holding		Whole-time	53	
		Part-time	54	
Wives or husbands of other partners and directors - if working on the holding			55	
Salaried managers			56	
Other family workers (see enclosed NOTES FOR GUIDANCE)	Regular whole-time	Male	57	
		Female	58	
	Regular part-time	Male	59	
		Female	60	
Hired workers (see enclosed NOTES FOR GUIDANCE)	Regular whole-time	Male	61	
		Female	62	
	Regular part-time	Male	63	
		Female	64	
Seasonal or casual workers - hired or family		Male	65	
		Female	66	
TOTAL Farmers, Growers and Workers			69	

YOUTH TRAINING

	Item	no. of trainees
Persons engaged by you as trainees under an official scheme and not paid AWB rates or more	48	

LIVESTOCK

- see enclosed NOTES FOR GUIDANCE

CATTLE AND CALVES

					number
Cows and heifers in milk	Mainly for producing milk or rearing calves for the **dairy herd**			70	
	Mainly for rearing calves for **beef**			71	
Cows in calf but not in milk	Intended mainly for producing milk or rearing calves for the **dairy herd**			72	
	Intended mainly for rearing calves for **beef**			73	
Heifers in calf (first calf)	Intended mainly for producing milk or rearing calves for the **dairy herd**	2 years & over		74	
		Under 2 years		75	
	Intended mainly for rearing calves for **beef**	2 years & over		76	
		Under 2 years		77	
Bulls for service	2 years old and over			78	
	1 year old and under 2 years			79	
All other cattle and calves	**2 years old and over**	Male - *exclude* bulls for service		80	
		Female	Intended for slaughter	81	
			For dairy herd replacements	94	
			For beef herd replacements	95	
	1 year old and under 2 years	Male - *exclude* bulls for service		83	
		Female	Intended for slaughter	84	
			For dairy herd replacements	85	
			For beef herd replacements	86	
	6 months old and under 1 year	Male - *include* bull calves for service		87	
		Female		88	
	Under 6 months old	Intended for slaughter as calves		89	
		Others	Male - *include* bull calves for service	90	
			Female	91	
TOTAL Cattle and calves				92	

		tick
Please tick this box if ALL the cattle entered at 92 above belong to someone else, and you are only providing grazing	93	

HORSES AND PONIES

		number
Horses and ponies owned by the occupier or occupier's family	125	
Horses and ponies **not** owned by the occupier or occupier's family	131	
TOTAL Horses and ponies	132	

GOATS

			number
Breeding females	Dairy	139	
	Non Dairy	142	
Other goats and kids		143	
TOTAL Goats		144	

FARMED DEER

		number
Farmed Deer (see enclosed NOTES FOR GUIDANCE)	96	

PIGS - weights are liveweight

			number
Breeding pigs	**Sows in pig**	100	
	Gilts in pig	101	
	Other sows - either being suckled or dry sows being kept for further breeding	102	
	Boars being used for service	103	
	Gilts 50 kg and over not yet in pig but expected to be used or sold for breeding	104	
Barren sows for fattening		105	
All other pigs (not entered above)	110 kg and over	106	
	80 to under 110 kg	107	
	50 to under 80 kg	108	
	20 to under 50 kg	109	
	Under 20 kg - *include* suckling pigs/piglets	110	
TOTAL Pigs		111	

SHEEP AND LAMBS

			number
Ewes, Shearlings and ewe lambs tupped/mated to produce lambs between 1st June 95 and 2nd June 96. (Number at 3rd June 96)	Intended to be retained or sold for further breeding	113	
	Intended for slaughter	116	
Female sheep not yet used for breeding, already put or to be put to the ram in 1996	1 year and over	114	
	under 1 year	112	
Rams and ram lambs used or to be used for service in 1996		115	
Other sheep 1 year and over		117	
Other lambs under 1 year old (exclude those recorded at 112 or 115)		118	
TOTAL Sheep and lambs		119	

FOWLS - *do not include* the same birds under more than one heading and *exclude* game birds

				number
Hens and pullets kept mainly for producing eggs for eating	**Growing pullets** - from day old to point of lay		121	
	Birds in the laying flock	Pullets from point of lay in first laying season	123	
		Hens (moulted) - *include* those in moult	124	
Fowls for breeding	**Breeding hens** (all ages) **laying eggs**	to hatch layer chicks	133	
		to hatch table chicks	134	
	Cocks and cockerels of all ages kept for breeding		126	
Table chicken under 7 weeks			127	
Table chicken 7 weeks and over			128	
TOTAL Fowls			137	

OTHER POULTRY

		number
Ducks of all ages	129	
Geese of all ages	130	
Turkeys of all ages	135	
All other poultry - *include* guinea fowl, ostriches etc.	138	

HORTICULTURE (enter all areas to the nearest 0.1 hectare)

- *include* crops against the principal item if named, or at item 200, 225 or 235 if the crop is not separately named or the area of each individual crop is less than 500 square metres
- see general notes for crops in enclosed NOTES FOR GUIDANCE

VEGETABLES GROWN IN THE OPEN FOR HUMAN CONSUMPTION

- *include* land rented out to processors etc.

			hectares
Brussels Sprouts		170	•
Cabbage - summer and autumn		172	•
All other cabbage - *include* spring cabbage		173	•
Cauliflower - summer and autumn maturing only. *Exclude* crops over-wintered in the field for spring harvest; *include* winter cauliflowers in item 200.		174	•
Calabrese - green sprouting broccoli often marketed as broccoli. *Exclude* over-wintered purple and white sprouting broccoli.		175	•
Carrots		178	•
Parsnips		181	•
Beetroot - red beet, not sugar beet or fodder beet		182	•
Onions	For salad	185	•
	Dry bulb - *include* previous autumn plantings	186	•
Broad beans		187	•
Runner beans - both pinched and climbing		190	•
French beans		192	•
Peas for harvesting dry			Enter in item 27
Green peas for fresh market		195	•
Vining peas for processing e.g. freezing, canning		196	•
Self blanching field celery		197	•
Lettuce - not under glass		198	•
Sweetcorn		199	•
All other vegetables- *include* watercress and rhubarb here, also *include* mixed areas (see enclosed NOTES FOR GUIDANCE). Enter total area in box 200 and specify each crop and its area below:-		200	•
TOTAL Vegetables grown in the open		201	•

BULBS AND FLOWERS GROWN IN THE OPEN

		hectares
TOTAL Bulbs and flowers grown in the open	244	•

JANUARY VEGETABLES, FLOWERS & BULBS

		yes/no
Do you expect to have more than 2 hectares of vegetables, flowers and/or bulbs in the 1996/97 growing season? Please answer YES or NO.	248	

ORCHARDS

		hectares
Orchards, not grown commercially - *include* orchards from which no fruit is sold for any purpose	207	•
Orchards, grown commercially - *include* orchards from which fruit is sold, including, pick your own sales and sales for juicing or other processing purposes. - *include* orchards of young, non-bearing trees, but not fruit stock - see item 230	208	•

SMALL FRUIT AND GRAPES

Strawberries	218	•
Raspberries	220	•
Blackcurrants	221	•
Gooseberries	223	•
Blackberries	217	•
Wine grapes	224	•
Other small fruit	225	•
TOTAL Orchards, small fruit and grapes (items 207 to 225 above)	226	•

HARDY NURSERY STOCK

			hectares
Fruit trees, bushes and canes, strawberries for runner production and other fruit stock for transplanting		230	•
Field grown	**Roses** - *include* stock for budding	231	•
	Shrubs, conifers, hedging plants and Christmas trees - not roses	232	•
	Ornamental trees and trees for amenity purposes	233	•
	Perennial herbaceous plants - not for cut flowers	234	•
Other hardy nursery stock and mixed areas - *include* land used for container-grown plants		235	•
TOTAL Hardy nursery stock		236	•

TOTAL HORTICULTURAL CROPS

- *exclude* mushrooms

		hectares
Sum of items 201+ 205+ 244+ 226+ 236 (to agree with item 22 on page 2)	249	•

PRODUCTION OF CONTAINER GROWN NURSERY STOCK

				number
In the last 12 months, how many plants were produced	**For sale in final pots of:**	less than one litre	237	
		1 to under 2 litres	238	
		2 to under 4 litres	239	
		4 litres and over	245	
	For growing on as liners		246	
TOTAL Container grown nursery stock Note: area occupied by this item to be *included* in 235 above			247	

GLASSHOUSE AND PROTECTED CROPS

- GLASSHOUSE AND PLASTIC COVERED STRUCTURES

NOTE A

- *Include* any fixed or mobile structure of a height sufficient to allow persons to enter in an upright position and which is glazed or clad with glass, rigid plastic, film plastic or other glass substitutes.
  In the case of mobile structures return only the area covered by the structures themselves and not the total area of the sites that could be covered by moving the structures.
  Give the total area of glasshouse floor space, not the area of benches or beds.
- *Include* area of bedding plants and plants in propagation for growing on or for sale to growers and gardeners in item 270
- *Exclude* crops under lights and cloches or low plastic tunnels.

AREA OF CROPS AT 3 JUNE - *exclude* crops under lights or cloches

			square metres
Area used for vegetables and fruit (Exclude vegetable and fruit plants in propagation for growing on or for sale to growers and gardeners. These should be included in item 270 below.)		260	
Area used for flowers and foliage for cutting and all other plants		270	
Remaining glasshouse area at 3 June	Area which you expect to crop in 1996	272	
	Area which you do not expect to crop in 1996	273	
TOTAL Area of glasshouses and plastic covered structures		274	
Glass and plastic covered structures erected or demolished since 1 June 1995	Erected	275	
	Demolished	276	
Divide item 274 by 10,000 to give glasshouse and protected area in hectares		205	Hectares •

NOTE B
The land on which the sheds or buildings stand should be returned at item 9 - other land.

MUSHROOMS - grown as a protected crop

		yes/no
Have you grown, or do you expect to grow, any mushrooms in 1996? Please answer YES or NO.	278	

TENANCY AND OWNERSHIP

OF THE TOTAL AREA (box 1 on page 2) - *exclude* seasonally let land

			hectares
How much is rented in by you?	Full Agricultural Tenancy (Agricultural Holdings Act **1986**)	411	•
	Farm Business Tenancy (Agricultural Tenancies Act **1995**)	412	•
	Other arrangement	413	•
How much is owned by you?		3	•

SEASONAL USE OF LAND

			hectares
Let **out** for 364 days or less **TO ANOTHER PERSON** for cropping, hay-making or grazing. (This land should be *included* in items 1 to 37)	Farm Business Tenancy (Agricultural Tenancies Act **1995**)	414	•
	Other arrangement	415	•
Let **in** for 364 days or less **FROM ANOTHER PERSON** for cropping, hay making or grazing. (*Do not include* this land in items 1 to 37)	Farm Business Tenancy (Agricultural Tenancies Act **1995**)	416	•
	Other arrangement	417	•

NOTE

If you have arranged to let out land for more than 364 days under the Agricultural Tenancies Act 1995, please remember to complete the LAND GIVEN UP section on page 6.

CHANGE IN AREA OF HOLDING

- **If your current total area is different** from the area printed on the front of the form (page 1, box 166) then, please -
  a) enter the correct total area in box 169 (on page 1)
  b) account for the difference by entering the changes below using a separate sheet if necessary
- **SEASONALLY LET LAND** should **NOT** be recorded as being given up or taken over

LAND GIVEN UP

		day	month	year
Date of change	292			

Show day, month, year as two digit numbers e.g. 3 June 1996 = 0 3 0 6 9 6

				hectares
Area of land given up	For non-agricultural use		294	•
	For farming by someone else	Farm Business Tenancy (Agricultural Tenancies Act 1995)	421	•
		Other leasing arrangement	422	•
		Sold or transferred	423	•

			hectares
Land given up officially designated as Less Favoured	Area of severely disadvantaged land	295	•
	Area of disadvantaged land	296	•

Holding No. If Known ..../........../..........

Name and address of new occupier ....

....

....

.... Postcode ....

LAND TAKEN OVER

		day	month	year
Date of change	282			

Show day, month, year as two digit numbers e.g. 3 June 1996 = 0 3 0 6 9 6

			hectares
Area of land taken over	Previously farmed by another person	281	•
	From non-agricultural use	284	•

			hectares
Land taken over officially designated as Less Favoured	Area of severely disadvantaged land	285	•
	Area of disadvantaged land	286	•

Holding No. If Known ..../........../..........

Name and address of previous occupier ....

....

....

.... Postcode ....

CHANGE OF NAME AND/OR ADDRESS

Please give any necessary correction to the name shown on page 1 in BLOCK LETTERS

Title	901		Initials	902
Surname	903			

Please give any necessary correction to the address shown on page 1 in BLOCK LETTERS

Address	904	
	905	
	906	
	907	
Postcode	908	

OTHER HOLDINGS IN THE SAME OCCUPANCY

Please list here any other holding reference numbers under which you make agricultural census returns

........../........../..........

........../........../..........

........../........../..........

........../........../..........

DECLARATION

- **Before signing this declaration please check that the appropriate entries have been made on all pages.**

I declare the particulars given in this return to be correct to the best of my knowledge and belief.

Signature of Occupier ....

Name (PLEASE PRINT) ....

Telephone Number .... Date 1996

The present occupier should complete and sign this return even if he or she has only recently taken over the holding and the name given on the form is not his or hers. If an accredited agent signs the form the name of the occupier should also be shown.

IN CONFIDENCE

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD

Agricultural and Horticultural Census: Return for 1 June 1993

Notice requiring information to be completed and sent back by 8 June 1993

I

Agricultural Census Branch
Government Buildings (Block A)
Epsom Road, Guildford, Surrey GU1 2LD
Tel: Guildford (0483) 68121

Under the Agricultural Statistics Act 1979 (as amended by the Agriculture (Amendment) Act 1984), the Minister of Agriculture, Fisheries and Food requires you to complete this form in respect of the land you occupy. **This is a legal requirement.** Under Section 4 of the 1979 Act, penalties may be imposed on any person who knowingly or recklessly gives false information, or who without reasonable excuse fails to provide information.

Notes for your guidance are provided on page one of the enclosed extra copy of the form which is for you to keep. Please read these notes carefully before you complete the form.

The information you give should relate to the position on **1 June 1993** except where otherwise stated on the form.

Please return the form by **8 June 1993** in the enclosed reply paid envelope.

No information you give on the form may be published or otherwise disclosed without your prior written consent, except as specified in Section 3 of the 1979 Act.

D E BRADBURY, Chief Statistician

In correspondence please quote your holding number

FOR OFFICIAL USE ONLY

B.R.	163	
I.C.	164	
L.S.	165	
M	167	
I.M.	168	

↑

166 Total recorded area shown above (hectares) (1 hectare = 2.471 acres)

PLEASE CHECK THESE IMPORTANT POINTS FIRST

POSTCODE Please enter correct postcode here if none is given on the address label above or if it is incorrect. 908

ACRES/ SQUARE FEET Please complete the form in hectares and square metres if possible. However if you wish to complete in acres and square feet please do so and tick this box. Whichever system is used **must be kept to throughout.** tick

TOTAL AREA If the **total area** of your holding is not as printed on the address label above please give the correct area in box 169 opposite and account for the difference on page 6. Do not include land on which the keep is let to you on a seasonal basis - see notes on page one of the yellow copy. 169 hectares •

HELP If you need any help with completion of this form, please write to the above address, (or telephone Guildford (0483) 68121) quoting your holding number, or consult your Regional Service Centre.

RETENTION COPY Also enclosed is a retention copy of this form. Census forms are kept for a limited period and requests for copies cannot normally be met; you may find it useful to complete and hold your retention copy.

AFTER COMPLETING THE FORM PLEASE SIGN THE DECLARATION AT THE FOOT OF PAGE 6

CSS/077

AREA OF HOLDING AND MAIN LAND USES

- see notes on page one of the yellow copy
- enter all areas to the nearest 0.1 hectare

CROPS AND FALLOW

			Item	hectares
Cereals for threshing	Wheat		11	•
	Barley	Winter Barley	12	•
		Spring Barley	13	•
	Oats		14	•
	Mixed corn		15	•
	Rye		16	•
	Triticale		33	•
Maize for threshing or stockfeeding			17	•
Potatoes - early and maincrop			19	•
Sugar beet not for stockfeeding			20	•
Hops			21	•
Horticultural crops - excluding mushrooms (to agree with item 249 on page 4)			22	•
Field beans			23	•
Peas for harvesting dry - human consumption or stockfeed			27	•
Other crops for stockfeeding	Turnips and swedes		24	•
	Fodder beet and mangolds		25	•
	Kale, cabbage, savoy, kohl rabi and rape		26	•
	Other crops - not grass Please specify:		28	•
Rape grown for oilseed			29	•
Linseed			30	•
Other crops not for stockfeeding (see notes) Please specify			31	•
Bare Fallow - do not include Set-Aside Land			32	•
TOTAL Crops and bare fallow			35	

GRASSLAND AND ROUGH GRAZING

- include clover, sainfoin and lucerne

	Item	hectares
Grassland put down in **1989** or later	5	•
All other grassland excluding rough grazing	6	•
Rough grazing on which you have **sole** grazing rights (see notes). Put grazed woodland in item 8.	7	•

WOODLAND

	Item	hectares
Woodland including grazed woodland on the holding	8	•

SET-ASIDE SCHEMES

	Item	hectares
Set-Aside Land. Land set-aside under an official payment scheme	34	•

ALL OTHER LAND

	Item	hectares
All other land not included above, e.g. farm roads, yards, buildings, gardens, ponds, derelict land	9	•

		Item	hectares
TOTAL Area of your holding (to agree with sum of items 35, 5 to 7, 8, 34 and 9 above)		1	•
Of the above area	How much is **RENTED** by you? - do not include seasonally rented land (see item 42)	2	•
	How much is **OWNED** by you?	3	•

GRASS GROWN FOR SEED

	Item	hectares
Area of grass already included in items 5 and 6 expected to be harvested for **seed** this year	40	•

IRRIGATION - do not include watercress

	Item	hectares
Total area of outdoor crops on your holding irrigated during 1992 season	43	•

SEASONAL USE OF LAND

	Item	hectares
Land currently let out for 364 days or less, **TO ANOTHER PERSON** for cropping, hay-making or grazing. (This land should be **INCLUDED** in items 1-35)	41	•
Land currently rented in for 364 days or less, **FROM ANOTHER PERSON** for cropping, hay-making or grazing. (**DO NOT INCLUDE** this land in items 1-35)	42	•

PERSONS WORKING ON THE HOLDING

- see notes on page one of the yellow copy

FARMERS, GROWERS AND WORKERS

- include each person once only. **Include** persons engaged by you as trainees under an official scheme **only if** they are paid AWB rates or more - else see item 48 below

			Item	number
Principal farmer/grower or partner - if working on the holding	Whole-time		50	
	Part-time		51	
Wife or husband of principal farmer/grower or partner - if working on the holding			52	
Other partners and directors - if working on the holding	Whole-time		53	
	Part-time		54	
Wives or husbands of other partners and directors - if working on the holding			55	
Salaried managers			56	
Other family workers (see notes)	Regular whole-time	Male	57	
		Female	58	
	Regular part-time	Male	59	
		Female	60	
Hired workers (see notes)	Regular whole-time	Male	61	
		Female	62	
	Regular part-time	Male	63	
		Female	64	
Seasonal or casual workers - hired or family		Male	65	
		Female	66	
TOTAL Farmers, Growers and Workers			69	

YOUTH TRAINING

	Item	no. of trainees
Persons engaged by you as trainees under an official scheme and not paid AWB rates or more	48	

LIVESTOCK

- see notes on page one of the yellow copy

CATTLE AND CALVES

					number
Cows and heifers in milk	Mainly for producing milk or rearing calves for the **dairy herd**			70	
	Mainly for rearing calves for **beef**			71	
Cows in calf but not in milk	Intended mainly for producing milk or rearing calves for the **dairy herd**			72	
	Intended mainly for rearing calves for **beef**			73	
Heifers in calf (first calf)	Intended mainly for producing milk or rearing calves for the **dairy herd**	2 years & over		74	
		Under 2 years		75	
	Intended mainly for rearing calves for **beef**	2 years & over		76	
		Under 2 years		77	
Bulls for service	2 years old and over			78	
	1 year old and under 2			79	
All other cattle and calves	2 years old and over	Male - excluding bulls for service		80	
		Female	Intended for slaughter	81	
			For dairy herd replacements	94	
			For beef herd replacements	95	
	1 year old and under 2	Male - excluding bulls for service		83	
		Female	Intended for slaughter	84	
			For dairy herd replacements	85	
			For beef herd replacements	86	
	6 months old and under 1 year	Male - including bull calves for service		87	
		Female		88	
	Under 6 months old	Intended for slaughter as calves		89	
		Others	Male - including bull calves for service	90	
			Female	91	
TOTAL Cattle and calves				92	

		tick
Please tick this box if **all** the cattle entered at 92 above belong to someone else, and you are only providing grazing	93	

HORSES AND PONIES

		number
Horses and ponies owned by the occupier or occupier's family	125	
Horses and ponies not owned by the occupier or occupier's family	131	
TOTAL Horses and ponies	132	

GOATS

			number
Breeding females	Dairy	139	
	Non Dairy	142	
Other goats and kids		143	
TOTAL Goats		144	

FARMED DEER

		number
Farmed Deer (see notes on page one of the yellow copy)	96	

PIGS - weights are liveweight

			number
Breeding pigs	**Sows in pig**	100	
	Gilts in pig	101	
	Other sows - either being suckled or dry sows being kept for further breeding	102	
	Boars being used for service	103	
	Gilts 50 kg (110 lb) and over not yet in pig but expected to be used **or sold** for breeding	104	
Barren sows for fattening		105	
All other pigs (not entered above)	110 kg (240 lb) and over	106	
	80 kg (175 lb) and under 110 kg (240 lb)	107	
	50 kg (110 lb) and under 80 kg (175 lb)	108	
	20 kg (45 lb) and under 50 kg (110 lb)	109	
	Under 20 kg (45 lb)	110	
TOTAL Pigs		111	

SHEEP AND LAMBS

		number
Ewes kept for breeding - do not include two-tooth ewes (item 114), or draft and cast ewes (item 116)	113	
Two-tooth ewes (shearling ewes or gimmers) put, or to be put, to the ram in 1993	114	
Rams for service	115	
Draft and cast ewes (do not include at item 113)	116	
Wethers and other sheep	117	
Lambs under 1 year old	118	
TOTAL Sheep and lambs	119	

FOWLS - do not include the same birds under more than one heading and do not include game birds

				number
Hens and pullets kept mainly for producing eggs for eating	**Growing pullets** - from day old to 18 weeks of age		121	
	Birds in the laying flock	Pullets over 18 weeks of age in first laying season	123	
		Hens (moulted) including those in moult	124	
Fowls for breeding	**Breeding hens (all ages) laying eggs**	to hatch layer chicks	133	
		to hatch table chicks	134	
	Cocks and cockerels of all ages kept for breeding		126	
Table chicken under 7 weeks			127	
Table chicken 7 weeks and over			128	
TOTAL Fowls			137	

TURKEYS

		yes/no
If you know you will keep turkeys on your holding in the next 12 months or think you are likely to do so, please answer YES. If not, put NO.	135	

OTHER POULTRY

		number
Ducks of all ages	129	
Geese of all ages	130	
All other poultry including turkeys and guinea fowl	138	

HORTICULTURE

- see notes on page one of the yellow copy
- enter all areas to the nearest 0.1 hectare

VEGETABLES GROWN IN THE OPEN FOR HUMAN CONSUMPTION

- include land rented out to processors etc.

			hectares
Brussels Sprouts		170	•
Cabbage - summer and autumn		172	•
All other cabbage including spring cabbage		173	•
Cauliflower - summer and autumn maturing only		174	•
Calabrese - green sprouting broccoli (often marketed as broccoli)		175	•
Carrots		178	•
Parsnips		181	•
Beetroot - red beet, not sugar beet or fodder beet		182	•
Onions	For salad	185	•
	Dry bulb - include previous autumn plantings	186	•
Broad beans		187	•
Runner beans	Pinched	189	•
	Climbing	190	•
French beans		192	•
Peas for harvesting dry			Enter in item 27
Green peas for fresh market		195	•
Vining peas for processing e.g. freezing, canning		196	•
Field celery self blanching - excluding wide row main crop		197	•
Lettuce - not under glass		198	•
Sweet corn		199	•
All other vegetables Include watercress and rhubarb here, also include mixed areas (see notes) Please specify		200	•
TOTAL Vegetables grown in the open		201	•

GLASSHOUSE AREA (1,000sq. metres = 0.1 hectare)

		hectares
Total area under glass or plastic structures **excluding** lights, cloches and low plastic tunnels (see note A on page 5)	205	•

BULBS AND FLOWERS GROWN IN THE OPEN

		hectares
Bulbs, corms, tubers and rhizomes for cut flowers or bulbs	240	•
Chrysanthemums	242	•
All other flowers for cutting	243	•
TOTAL Bulbs and flowers grown in the open	244	•

OCTOBER VEGETABLES AND BULBS

		yes/no
Do you expect to have more than 0.5 hectare of vegetables and/or bulbs in October this year?	248	

ORCHARDS

				hectares
Orchards, not grown commercially			207	•
Orchards, grown commercially - include area of young non bearing orchards but not fruit stock (item 230)	**Dessert apples**	Cox's Orange Pippin and other Cox clones	208	•
		All other varieties	209	•
	Cooking apples	Bramley's Seedling	210	•
		All other varieties	211	•
	Cider apples and perry pears		212	•
	Pears		213	•
	Plums		214	•
	Cherries		215	•
	Other top fruit - including nuts		216	•

SMALL FRUIT AND GRAPES

Strawberries	Open grown only	218	•
	Under cloches or low tunnels	219	•
Raspberries		220	•
Blackcurrants	For market	221	•
	For processing	222	•
Gooseberries		223	•
Blackberries		217	•
Wine grapes		224	•
Other small fruit		225	•
TOTAL Orchards, small fruit and grapes (items 207 to 225 above)		226	•

HARDY NURSERY STOCK

			hectares
Fruit trees, bushes and canes, strawberries for runner production and other fruit stock for transplanting		230	•
Field grown	**Roses** - including stock for budding	231	•
	Shrubs, conifers, hedging plants and Christmas trees - not roses	232	•
	Ornamental trees	233	•
	Perennial herbaceous plants - not for cut flowers	234	•
Other hardy nursery stock and mixed areas - include land used for container-grown plants		235	•
TOTAL Hardy nursery stock		236	•

		hectares
TOTAL Horticultural crops - excluding mushrooms. Sum of items 201+ 205+ 244+ 226+ 236 (to agree with item 22 on page 2)	249	•

PRODUCTION OF CONTAINER GROWN NURSERY STOCK

				number
In the last 12 months, how many plants were produced	**For sale in final pots of:**	0.9 litres or less	237	
		1 - 1.9 litres	238	
		2 - 4.0 litres	239	
		More than 4 litres	245	
	For growing on as liners		246	
TOTAL Container grown nursery stock Note: area occupied by this item to be included in 235 above			247	

GLASSHOUSE AND PROTECTED CROPS

- GLASSHOUSE AND PLASTIC COVERED STRUCTURES
  1,000 square feet = 93 square metres

NOTE A
Include any fixed or mobile structure of a height sufficient to allow persons to enter in an upright position and which is glazed or clad with glass, rigid plastic, film plastic or other glass substitutes. In the case of mobile structures return only the area covered by the structures themselves and not the total area of the sites that could be covered by moving the structures.
Do not include lights and cloches or low plastic tunnels.

TOTAL AREA (whether in use or not - see note A)

			square metres
Area covered by glass	With heating equipment	250	
	Without heating equipment	251	
Area covered by plastic or other glass substitutes	With heating equipment	252	
	Without heating equipment	253	
TOTAL Area of glasshouses and plastic covered structures (to agree with item 274 below)		254	

CHANGE OF AREA

			square metres
Glass and plastic covered structures erected or demolished since 1 June 1992	Erected	275	
	Demolished	276	

AREA OF CROPS AT 1 JUNE
- do not include crops under lights or cloches

					square metres
Vegetables - excluding plants in propagation (see item 266)	Tomatoes	Heated crop	Planted by 28th February 1993	255	
			Planted after 28th February 1993	256	
		Unheated crop		257	
	Cucumbers			258	
	Sweet peppers			259	
	Lettuce			267	
	Other vegetables and herbs			260	
Flowers and foliage for cutting	Pinks			277	
	Carnations - excluding pinks			261	
	Alstroemeria			269	
	Roses			262	
	Chrysanthemums - excluding pots (see item 268)			263	
	Other flowers and foliage - exc. plants for sale or in propagation			270	
Plants (see note B)	For sale as pot plants	Chrysanthemums		268	
		Other flowering plants		264	
		Foliage plants		265	
	Plants in propagation for growing on or for sale to growers and gardeners (see note C)			266	
Fruit	Strawberries and any other fruit (see note D)			271	
Remaining glasshouse area at 1 June	Area which you expect to crop in 1993			272	
	Area which you do not expect to crop in 1993			273	
TOTAL Crops and remaining glasshouse area (to agree with total at 254 above)				274	

NOTE B
For items 268 and 264 to 266 give the total area of glasshouse floor space, not the total area of benches or beds. **Include** vegetables for commercial production at item 266.

NOTE C
Include present area of bedding plants at item 266. Information is **not** required on numbers produced.

NOTE D
Enter strawberries grown in the open under cloches or low tunnels at items 218 or 219.

MUSHROOMS - grown as a protected crop

		pounds (lbs)
In the last 12 months what was your total production? (see note E)	278	

NOTE E
The land on which the sheds or buildings stand should be returned at item 9 - other land.

CHANGE IN AREA OF HOLDING

- **If your current total area is different** from the area printed on the front of the form (page 1, box 166) then, please -
  a) enter the correct total area in box 169 (on page 1)
  b) account for the difference by entering the changes below

- **SEASONALLY LET LAND** should **NOT** be recorded as being given up or taken over

LAND GIVEN UP

		day	month	year
Date of change	292			

Show day, month, year as two digit numbers e.g. 1 June 1992 = 0 1 0 6 9 2

			hectares
Area of land given up	For farming by another person	291	•
	For non-agricultural use	294	•

		hectares
Land given up officially designated as Less Favoured	Area of severely disadvantaged land	•
	Area of disadvantaged land	•

Name and address of new occupier

....

....

....

....

.... Postcode ....

LAND TAKEN OVER

		day	month	year
Date of change	282			

Show day, month, year as two digit numbers e.g. 1 June 1992 = 0 1 0 6 9 2

			hectares
Area of land taken over	Previously farmed by another person	281	•
	From non-agricultural use	284	•

		hectares
Land taken over officially designated as Less Favoured	Area of severely disadvantaged land	•
	Area of disadvantaged land	•

Name and address of previous occupier

....

....

....

....

.... Postcode ....

CHANGE OF NAME AND/OR ADDRESS

Please give any necessary correction to the name shown on page one in BLOCK LETTERS

Title	901		Initials	902
Surname	903			

Please give any necessary correction to the address shown on page one in BLOCK LETTERS

Address	904	
	905	
	906	
	907	
Postcode	908	

OTHER HOLDINGS IN THE SAME OCCUPANCY

Please list here any other holding reference numbers under which you make agricultural census returns

.... / .... / ....

.... / .... / ....

.... / .... / ....

.... / .... / ....

OTHER LIVESTOCK ON YOUR HOLDING

Are there any livestock (of the types specified on page 3) on this holding which you have not included in your return for any reason? yes/no

If **'Yes'** please give details:

....

....

....

....

DECLARATION

I declare the particulars given in this return to be correct to the best of my knowledge and belief.

Signature of Occupier ....

Name (PLEASE PRINT) ....

Date 1993

Telephone Number ....

The present occupier should complete and sign this return even if he or she has only recently taken over the holding and the name given on the form is not his or hers. If an accredited agent signs the form the name of the occupier should also be shown.

(36)

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD

Agricultural and Horticultural Census: Return for 14 March 1994

Notice requiring information to be completed and sent back by 21 March 1994

I

Agricultural Census Branch
Government Buildings (Block A)
Epsom Road, Guildford, Surrey GU1 2LD
Tel: Guildford (0483) 68121

Under the Agricultural Statistics Act 1979 (as amended by the Agriculture (Amendment) Act 1984), the Minister of Agriculture, Fisheries and Food requires you to complete this form in respect of the land you occupy. **This is a legal requirement.** Under Section 4 of the 1979 Act, penalties may be imposed on any person who knowingly or recklessly gives false information, or who without reasonable excuse fails to provide information. Please return the form by **21 March 1994** in the enclosed reply paid envelope.

This enquiry is made to obtain up to date statistics of agriculture and horticulture in England. The results will provide the government with the information necessary to formulate agricultural policy and to meet certain of the United Kingdom's obligations to the European Community. They are also used extensively by the industry itself.

The information you give should relate to the position on **14 March 1994** except where otherwise stated on the form.

No information you give on the form may be published or otherwise disclosed without your prior written consent, except as specified in Section 3 of the 1979 Act.

P F HELM, Chief Statistician

In correspondence please quote your holding number

FOR OFFICIAL USE ONLY

B.R.	163	
I.C.	164	
L.S.	165	
I.M.	168	
D	396	

↑

166 Total recorded area shown above in hectares (1 hectare = 2.471 acres)

PLEASE CHECK THESE IMPORTANT POINTS FIRST

COMPLETION	This form should be completed only in respect of the holding named above. If you occupy any other agricultural holdings please also tick this box and complete the section on other holdings on page 4.	tick 40
POSTCODE	If no postcode is shown on the address label above or if it is incorrect please enter the correct postcode in box 908 on page 4.	
ACRES/ SQUARE FEET	Please complete the form in hectares and square metres if possible. However if you wish to complete in acres and square feet please do so and tick this box. Whichever system is used **must be kept to throughout.**	tick
TOTAL AREA	If the **total area** of your holding is not as printed on the address label above please give the correct area in box 169 opposite and account for the difference on page 4. *Exclude* land on which the keep is let to you on a seasonal basis - see NOTES FOR GUIDANCE on page three.	hectares 169 •
HELP	Notes for guidance to help with completion of this form are on page 3. If you need any further help with completion of this form, please write to the above address, (or telephone Guildford (0483) 68121) quoting your holding number.	

AFTER COMPLETING THE FORM PLEASE SIGN THE DECLARATION AT THE FOOT OF PAGE 4

SEASONALLY LET LAND

- see section A of the 'Notes for Guidance' opposite

		tick
If all the land on your holding is seasonally let for 364 days or less to another person for grazing or cropping please tick this box.	41	

Please go on to complete the rest of the form giving your best estimate, taking these points into consideration.

- *include* land let under a seasonal grazing licence and any animals kept on this land.
- *exclude* land let on a full agricultural tenancy to another farmer and included in his/her return.
- *exclude* land rented by you for 364 days or less for grazing or cropping.

AREA OF HOLDING AND MAIN LAND USES

- enter all areas to the nearest 0.1 hectare
- Set-Aside land and crops grown on Set-Aside land should be entered under item 5 below
- see section A of the 'Notes for Guidance' opposite

LAND USE		hectares
Cereals	1	•
Potatoes	2	•
Horticultural crops - see section A of the 'Notes for Guidance' opposite	3	•
All other crops and bare fallow (*except* grass)	4	•
Set-Aside land & crops grown on Set-Aside land	5	•
All grassland (*except* rough grazing)	6	•
Rough grazing on which you have SOLE grazing rights	7	•
Woodland and other land not recorded above (*include* farm roads, buildings and ponds etc.)	8	•
TOTAL AREA (Items 1 to 8)	9	•

		hectares
Of the above total area how much is rented by you? (*exclude* seasonally rented land)	10	•

		square metres
Glasshouse area	11	

PERSONS WORKING ON THE HOLDING

- see section B of the 'Notes for Guidance' opposite
- *exclude* persons engaged by you as trainees under an official scheme and not paid AWB rates or more.

LABOUR			number
Farmers, growers, partners and directors (if working on the holding)	Whole-time	12	
	Part-time	13	
Wives or husbands of the above (if working on the holding)		14	
Regular whole-time workers (hired or family)		15	
Regular part-time workers (hired or family)		16	
Other workers on the holding (*include* seasonal and casual workers)		17	
TOTAL LABOUR (Items 12 to 17)		18	

LIVESTOCK

- see section C of the 'Notes for Guidance' opposite

CATTLE AND CALVES			number
Cows and heifers in milk and cows in calf but not in milk	Mainly for milk production	19	
	Mainly for beef production	20	
All other cattle and calves		21	
TOTAL Cattle and calves		22	

PIGS		number
Sows and gilts in pig and other sows being suckled or kept for further breeding	23	
All other pigs	24	
TOTAL Pigs	25	

SHEEP AND LAMBS		number
Ewes and lambs kept for breeding	26	
All other sheep and lambs	27	
TOTAL Sheep and lambs	28	

POULTRY		number
Fowls - hens and pullets kept mainly for producing eggs for eating (*include* growing pullets)	29	
All other poultry	30	
TOTAL Poultry	31	

GOATS			number
Breeding females	Dairy	32	
	Non - dairy	33	
All other goats (*include* kids)		34	
TOTAL Goats		35	

HORSES AND PONIES		number
Horses and ponies owned by the occupier or occupier's family	36	
Horses and ponies not owned by the occupier or occupier's family	37	
TOTAL Horses and ponies	38	

OTHER ACTIVITIES ON THE HOLDING

Please specify (e.g. mushrooms)

....

....

PLEASE GO ON TO COMPLETE PAGE 4 OVERLEAF

Appendix B

Publications

Dragosits U., Sutton M.A. and Place C.J. (1996b): The spatial distribution of ammonia emissions in Great Britain for 1969 and 1988 assessed using GIS techniques. In: Sutton M.A., Lee D.S., Dollard G.J. and Fowler D. [Eds.] (1996): Poster Proceedings: International Conference on Atmospheric Ammonia. Culham, Oxford 2-4 October 1995. Published: ITE (Edinburgh Research Station), Bush Estate, Penicuik, Midlothian EH26 0QB. p. 46-49.

Dragosits U., Sutton M.A., Place C.J., and Bayley A.A. (1998): Modelling the spatial distribution of ammonia emissions in the UK. *Environmental Pollution* **102**, 195-203.

Permission has been obtained from the joint authors and publishers to include a copy of the above listed papers.

ELSEVIER

Environmental Pollution 102, S1 (1998) 195–203

ENVIRONMENTAL POLLUTION

Modelling the spatial distribution of agricultural ammonia emissions in the UK

U. Dragosits[a,b]*, M.A. Sutton[a], C.J. Place[b] and A.A. Bayley[c]

[a]Institute of Terrestrial Ecology, Bush Estate, Penicuik, Midlothian EH26 0QB, Scotland, UK
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Abstract

Accurate models of the spatial distribution of ammonia (NH_3) emissions are an essential input to models of atmospheric transport and deposition. This is especially important when resulting deposition maps are used to calculate patterns of critical loads exceedance or to determine suitable abatement measures. A new methodology has been developed to model the distribution of agricultural ammonia emissions and is applied here for the UK. The model employs a specific spatial weighted redistribution of NH_3 emission sources onto suitable landcover types at a 1-km grid level. Key input data to the model are agricultural census data, a satellite-based landcover map and estimates of NH_3 emission source strength. The model provides more realistic spatial NH_3 emission estimates than previous models, especially for semi-natural/natural areas by relocating emission sources from extensively used upland areas to the more intensively farmed lowland areas within each parish. At present the model results are summarised as maps at a 5-km grid resolution to reduce uncertainty in the spatial location of NH_3 sources. Compared with coarser resolution estimates this also provides a more accurate link to critical load exceedances. The more accurate redistribution also reduces the apparent critical loads exceedance on upland areas. Results are presented and compared for 1988 and 1996. These show broadly similar patterns between the years, although substantial local changes have occurred, particularly for intensive livestock farming. The model has been used to generate initial spatially resolved abatement scenarios and provides a general tool for locating NH_3 emissions that could be applied to other regions.

Keywords: NH_3; Inventory; Mapping; GIS; Spatial modelling

Introduction

Atmospheric NH_3 emissions are extremely variable in space, both on a regional scale as well as on a local scale. The availability of reliable national NH_3 emissions maps at a suitable resolution is an essential prerequisite for atmospheric transport, nitrogen deposition and critical loads exceedance models (e.g. Singles et al., 1998; Sutton et al., 1998). Any policy formulated for abatement purposes ultimately depends on these data.

Ammonia emissions originate mainly from agricultural sources and their spatial distribution is therefore closely linked to agricultural production. This enables the development of spatial emission models based on annual agricultural census returns, land cover information and ammonia source strength estimates.

The spatial distribution of NH_3 emissions in the UK was first estimated by ApSimon et al. (1987) and Kruse et al. (1989) for England and Wales at a resolution of 10×10 km. More recently, Eager (1992), Sutton et al. (1995) and Dragosits et al. (1996) produced the first maps for Great Britain at a 5-km resolution for 1988. These maps were made using general methodologies, spatially locating the source items as such (e.g. cattle) rather than as ammonia emission sources (e.g. grazing, housing/storage and landspreading emissions from cattle redistributed according to their relative importance and spatial location as separate sub-sources).

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The main aim of this study was to define present spatial patterns of NH_3 emissions based on average agricultural practice in the United Kingdom and to produce more reliable maps than made using previous methodologies. A new methodology has been developed here, the work updated from 1988 to 1996 and Northern Ireland included for the first time. In the following, the available data sources are described briefly, the applied new methodology discussed and the results examined in detail.

Methods

General methodology

The main data sources for modelling NH_3 emissions in the UK are the annual June Agricultural and Horticultural Census, land cover data and NH_3 source strength estimates. In previous national NH_3 emission inventories, existing gridded 5-km census data were used. These data originate from a general spatial redistribution model for census data developed by the Edinburgh Data Library (Hotson 1986) at a 1-km grid resolution. The landcover data used in this approach date back to the late 1970s and are (a) assigned to one of two classes with agricultural potential or (b) excluded from agricultural use for each 1-km square. The model allows for the absence of a census item for any 1-km square with unfavourable landcover in the parish, so that the total is distributed evenly over the remaining area. This approach is reasonable in providing a broad picture of the distribution of census items across the country.

The purpose of the disaggregation of the parish census data is to produce best estimates of the likely spatial distribution of agricultural activities at a 5 km grid scale. However, the choice of rules governing the redistribution of census items in relation to land cover significantly influences the overall distribution of the results. This is because NH_3 emissions do not occur equally through all parts of livestock management. Emissions from livestock housing, storage and application of wastes are much larger than from grazing animals. Thus, although animals may graze hill land at some time, most of the NH_3 emissions will be located within better agricultural land at lower altitude.

For the purposes of an NH_3 emissions inventory, it is desirable to distinguish between intensively and extensively used agricultural areas, especially within the larger parishes. Such parishes, especially in the Highlands of Scotland, tend to have large areas of very extensively used land such as moorland. Using the general methodology (Hotson, 1986), any gridsquare in e.g. the moorland category would become populated with some livestock types at the same rate as other grid squares which would potentially be used for intensive agricultural activities. This would have the effect of reducing the estimated as compared with real concentration of livestock in the intensively used areas and increasing the apparent concentration in the extensively used areas. This aspect is especially important if nitrogen deposition and critical loads exceedance models are derived from the results of an NH_3 emissions inventory.

Data sources

Agricultural census data

For this study, agricultural census data were available as parish summary data for Great Britain for 1988 and as disclosive parish summary data for 1996, providing information on livestock numbers and crop areas for the holdings in each parish (for definitions of holdings, see MAFF et al., 1996). A condition for use of the disclosive parish census data for 1996 was that the model output resulted in non-disclosive ammonia-emission maps, i.e. that information given for any particular farm could not be identified.

Landcover data

A suitable national landcover dataset at a resolution of sub-1 km was identified in the ITE landcover data set, a classified satellite (LANDSAT Thematic Mapper) map of Great Britain (Barr et al., 1993). For each 1-km grid cell, percentage values for each of the 26 landcover types are available for a base year of 1990. These cover types were aggregated to selected six classes for the purpose of this study, arable land, good grassland, partially improved grassland, poor rough grassland, very poor rough grazing land (heather, etc.) and suburban/rural development.

Emission source strength data

Ammonia emissions per unit livestock or per hectare crop vary and are dependent on many environmental factors and differences in agricultural practice between farms (e.g. Jarvis and Pain, 1990). For instance, annual emissions from grazing livestock such as cattle or sheep vary greatly with the amount of nitrogen applied to pastures.

The greater the proportion of the year the animals spend outdoors grazing, the smaller the total annual emissions per animal are expected to be. This may partially depend on the potential maximum length of the grazing season, which in turn depends on climatic and topographic factors, but also on the husbandry regime a farmer chooses to apply etc. Some of these factors can be modelled to a certain degree, provided sufficient spatial data are available. In this study, average conditions over the whole country were assumed. Total annual NH_3

Table 1

Ammonia emission estimates for livestock classes in the UK 1996 (totals may not add up entirely due to rounding)

Category	Animal numbers (UK)	Emission/animal (kg NH_3–N year^{-1})	Total NH_3 –N (kt year^{-1})	Contribution (%)
Cattle	11,904,000	11.23	133.7	57.4
Sheep, goats	41,623,000	0.38	15.9	6.8
Pigs	7,506,000	3.18	23.9	10.3
Poultry	146,496,500	0.19	27.8	12.0
Horses	302,000	6.56	2.0	0.9
Deer	33,700	0.95	0.03	0.01
Total livestock	-	-	203.2	87.3
Crops & grassland	-	volatilisation: 2.94% (of N fertiliser applied)	29.7	12.7
Total emissions	-	–	232.9	100.0

Table 2

Proportions of NH_3 emission components for example livestock classes (derived from TFEI, 1996) (individual percentages may not add up entirely due to rounding)

%	Dairy cows	Fattening pigs	Sheep	Horses	Broilers
Housing + storage	44	58	18	37	60
Spreading	42	41	16	27	41
Grazing	14	0	65	37	0

emissions for each livestock type were derived from the official NH_3 emission figures (DoE, 1995) (Table 1).

In detail, ammonia emissions have been assigned to four main components:

- livestock housing and waste storage
- landspreading of livestock wastes (surface spreading of manure assumed)
- livestock grazing
- fertiliser application to agricultural and horticultural crops and grassland.

Ammonia emissions from each livestock type under average husbandry conditions have been apportioned to these different NH_3 emission components (Table 2).

Ammonia emissions from the application of mineral fertiliser to crops and grassland are dependent on the fertiliser type and N fertiliser application rate. The N fertiliser application rates typical for crops in Great Britain were available in detail for the main crops and crop groups from the British Survey of Fertiliser Practice for 1988 and 1996 (Dyer et al., 1989; The Stationery Office, 1997). An average fertiliser emission rate for different types of fertiliser was assumed to be evenly distributed over the country and all crops. An estimated average volatilisation factor of 2.94% of the applied mineral N fertiliser was derived from the previously agreed emission figures (DoE, 1995; see Dragosits et al., 1996).

Methodology for the spatial redistribution of emission sources

The new emissions model (Fig. 1) is based on the spatial redistribution of the census data over suitable landcover types as NH_3 emission sub-sources, rather than simply as general items. It also applies the same type of data and methods for all of Great Britain, so that discontinuities at the border between different countries are not an issue.

The four main livestock sources of NH_3 emissions tend to occur on specific landcover types, i.e. livestock housing and manure storage will be located close to or on the farm itself, livestock grazing will occur on grassland etc. Although the landcover types on the landcover map are not equivalent to land use (Wyatt et al., 1990) (e.g. what the satellite classification identifies as grassland, could be a pasture or a football pitch), a strong correlation can be observed and landuse can be inferred. Therefore, the four main components of agricultural activities can be linked to different landcover types for the accuracy required of a national inventory at a 5-km grid resolution.

For each parish, the livestock and crop items from the Census data were apportioned to the best suited landcover classes, with livestock items divided according to the percentage values as illustrated in Table 2. A weighted distribution approach dependent on the total area of each of the landcover classes available for agricultural use within the parish was taken to ensure realistic results. The main objective here was to distribute emission sources to where they were most likely located on the ground. For the distribution of grazing emissions, a sub-model was established to take stocking densities on different quality grazing land into account. For instance, in the rather large parishes in upland Britain an even distribution of animals over all land potentially used for grazing would result in a distorted spatial distribution of NH_3 emissions.

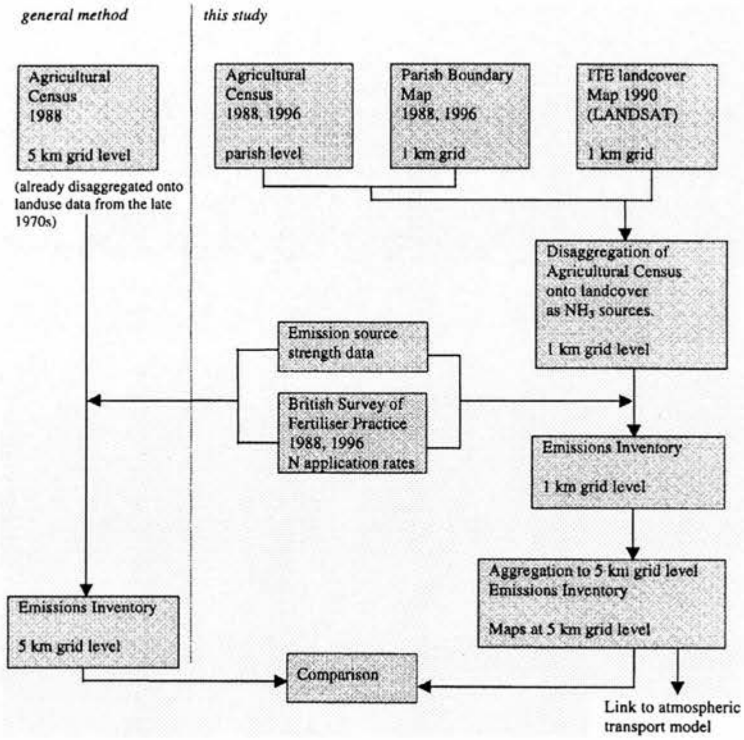

Fig. 1. Methodology for modelling of NH_3 emissions in Great Britain.

The model was implemented at the same resolution as that of the landcover data and parish boundary input data (1-km grid squares) to achieve the best possible spatial accuracy in locating census items for the emissions model. This was necessary to account for the irregular shape of most parishes, i.e. to minimize the error in converting parish data to the desired output level.

An example of the applied methodology is shown in Fig. 2 for beef cattle for an area of the Scottish/English borders, comparing the general redistribution by the Edinburgh Data Library with the new redistribution model based on a landcover dependent weighting for NH_3 emissions. The effect of the modified approach is a redistribution of census items as 'NH_3 sources' as opposed to census items as such. The differences between the two methods can be seen clearly in both the 1-km and 5-km maps. The parish boundaries, clearly visible in Fig. 2a, largely disappear in 2b. The revised approach shows a much greater concentration of emissions in the valleys, as would be expected. This trend is reflected in the aggregated 5-km grid maps. A discontinuity is also evident in the general methodology map (2a,c) between England and Scotland. This arises mainly from different re-allocation rules for beef cattle for the two countries in the general methodology and is avoided in the new approach.

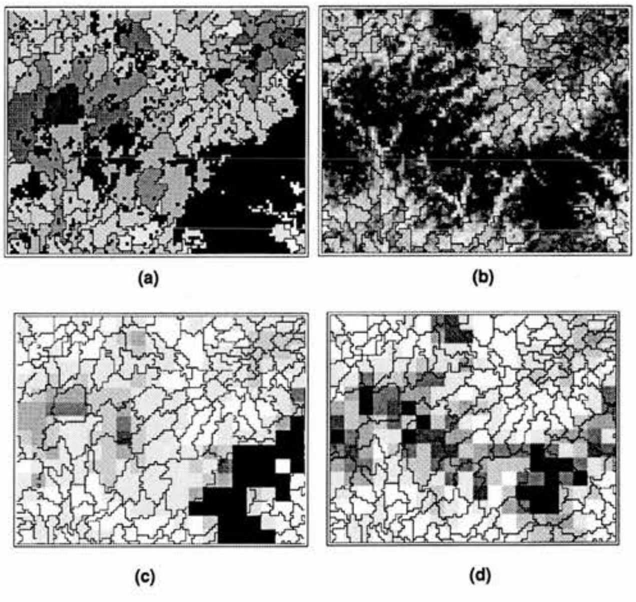

Fig. 2. Comparison between the general methodology and the new NH_3 model of redistributing beef cattle for an area of the England/Scotland borders. Lines represent 1-km resolution boundaries of parishes. White areas indicate high NH_3 emission and black areas no NH_3 emission. A discontinuity between England and Scotland can be seen in Fig. 2a and 2c, with England (mostly black) in the lower right hand corner.

Methodology for the calculation of ammonia emissions

After redistributing all census items as NH_3 sources over the landcover data, the calculation of emissions is a product of average emission source strength estimates (see Table 1) with matrices containing the number of sources, i.e. the number of animals in each livestock category and the total amount of mineral N fertiliser

Table 3

Ammonia emissions from agricultural sources in the UK 1988 and 1996

Category	Animal numbers (UK 1988)[b]	Animal numbers (UK 1996)[a]	Total NH_3– N (kt/ year) 1988	Total NH_3– N (kt /year) 1996
Cattle	11,902,000	11,904,000	133.7	133.7
Sheep, goats	41,028,000	41,623,000	15.6	15.9
Pigs	7,983,000	7,506,000	25.4	23.9
Poultry	132,866,000	146,496,500	25.2	27.8
Horses	n/a	302,000	2.0[c]	2.0
Deer	n/a	33,700	0.03[c]	0.03
Total livestock	–	–	202.0	203.2
Crops & grass	–	–	32.5[b]	29.7[a]
Total emissions	–	–	234.5	232.9

Sources: [a]UK Census 1988: MAFF et al. (1994); [b]UK Census 1996: sum of model input data; [c]estimated as equivalent to 1996 due to data unavailability in the Agricultural Census 1988 for England and Wales.

applied to crops, for each grid square in the model domain.

Once the methodology is in place, scenarios using different emission source strength estimates can be calculated. This includes, for instance, the estimation of reductions from abatement measures applied to certain categories of livestock. The present model can also be used to refine the emission estimates by taking subcategories of livestock into account such as a differentiation between dairy cows and beef cattle, or between laying hens, table chickens and other poultry (such as turkeys, geese and ducks).

It is intended to further improve the results at a later stage by introducing spatially variable emission source strength estimates. Examples for this would be the introduction of environmental data (climate, soil, etc.) and data regarding the spatial variation of fertiliser application rates to determine more localised emission source strength estimates. These matrices could then be used in the emissions model in conjunction with the matrices of redistributed census items. The methodology described in this subsection was applied to Great Britain only. For Northern Ireland, data were provided differently for 1996 and used directly as a 5×5 km grid data set, which was created by amalgamation of holdings data for each grid square.

Results and discussion

Comparison between results using the old and new methodology for 1988

The new methodology of spatially redistributing NH_3 emissions described above does not change the total sums for Great Britain (Fig. 3), since the emission source strength estimates per unit livestock or per kg N fertiliser per hectare are still applied as averages for the whole country (Table 1). It does, however, have a significant impact on the NH_3 emissions at any particular location. Ammonia emissions have been concentrated in areas which are more suitable for intensive agricultural activities within each parish. Conversely, areas with little agricultural activity such as moorland, heathland and other seminatural vegetation types have had NH_3 sources and therefore emissions reduced. This is due to the conditions and rules specified in the model which were developed to mirror the reality of agricultural practice.

Since the atmospheric lifetime of gaseous NH_3 is rather short, large gradients of NH_3 deposition occur downwind of sources on spatial scales of <10 km as well as at the field scale. Reasonably accurate spatial distributions of NH_3 emissions are especially important for defining source and sink areas regarding NH_3 deposition.

Ammonia emission inventories for 1988 and 1996

General trends

Total agricultural NH_3 emissions for the UK appear to have declined very slightly between 1988 and 1996, as shown in Table 3. This change is not readily identifiable as the fine detail of the totals in this table are not directly comparable, mainly due to changes in the Agricultural Census. As far as emissions from livestock are concerned, the total appears to have remained similar between 1988 and 1996, despite a decrease in emissions from pig farming and an increase in emissions from poultry farming (some of which is due to turkeys being included in the 1996 Census in England and Wales).

However, changes in the spatial distribution of emissions are more evident (see Fig. 3). It should be noted that any differences between 1988 and 1996 discussed below do not include Northern Ireland, as no spatial data were available for 1988.

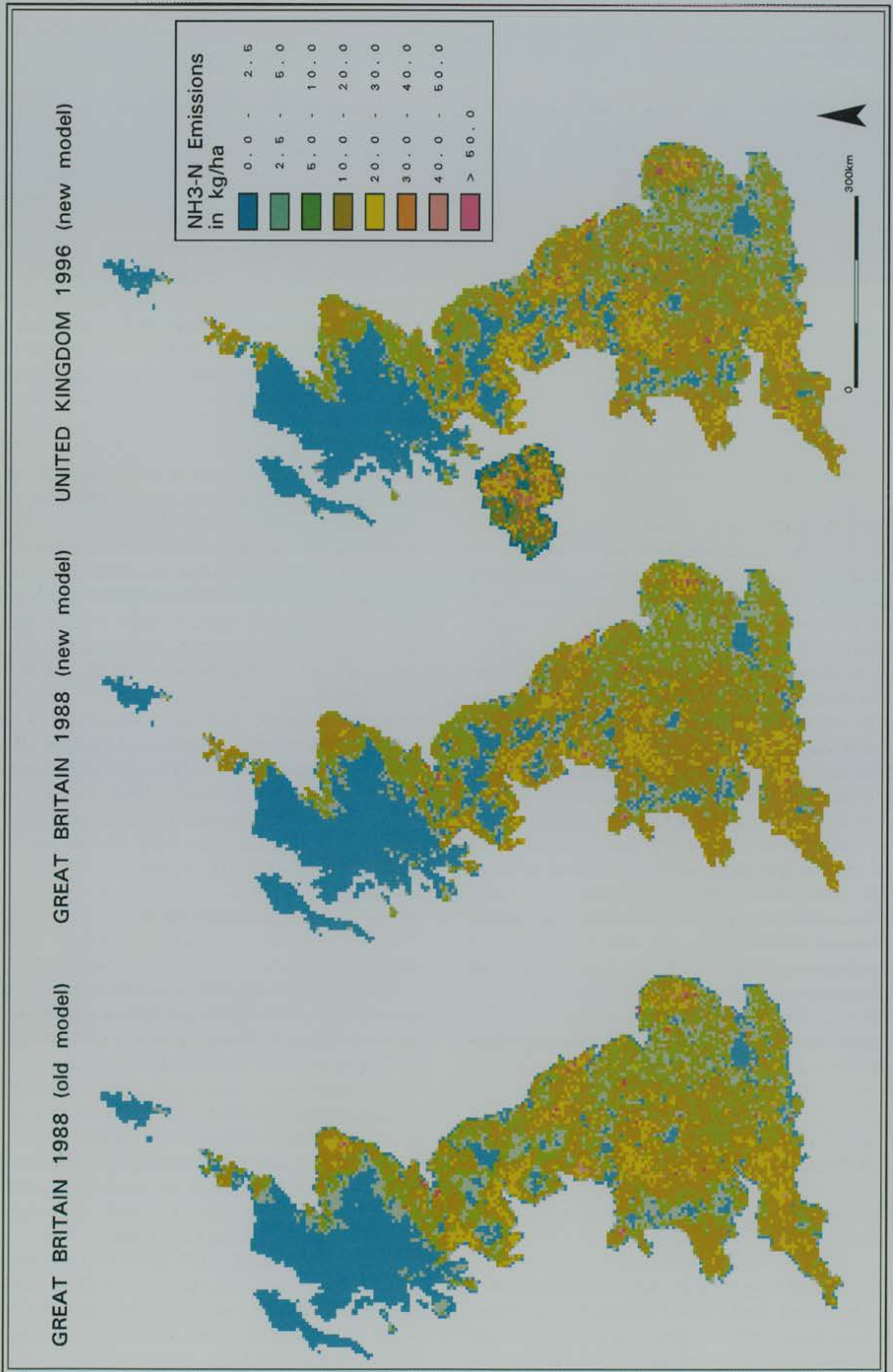

Fig. 3. Total agricultural ammonia emissions for (a) Great Britain, old methodology; (b) Great Britain 1988, new methodology; (c) United Kingdom 1996, new methodology.

Differences in the spatial patterns of emissions in Great Britain 1988/1996

The spatial differences between the two reference years were compared for each gridcell on a 1-km and 5-km model output level. For most areas, differences in total NH_3 emissions between 1988 and 1996 are quite small, with about a quarter of all 1-km and 5-km grid squares with less than ± 0.1 kg ha^{-1} difference (equivalent to ± 250 kg N for the whole 5-km grid square). About 60% of all 1-km and 5-km grid squares are within ± 1kg ha^{-1}. However, there is still a significant number of grid squares with very large changes (–60 to +80 kg ha^{-1} for the 5-km grid; –500 to +1700 kg ha^{-1} for the 1-km grid). These squares mostly represent areas with intensive livestock farming, the positive changes showing new developments, intensification or relocation since 1988, the negative changes representing disappearance, extensification or relocation. However, some of these changes may be artifacts: For 1996, it was possible to use potentially disclosive model input data, as long as non-disclosivity was ensured at the output level (5-km grid). It was therefore possible to assign parishes in disclosive output gridsquares to adjacent parishes rather than a discontinuous "left-over" parish. If there were large intensive livestock units present in these notional parishes for 1988, they were smoothed out over all parishes included into these summary parishes within each county.

The general pattern of change for Great Britain shows several significant trends. The larger positive and negative changes ($> \pm 5$ kg ha^{-1} in a 5-km square) only occur in a small number of grid squares and appear to be linked primarily to pig and poultry farming. Smaller changes between 1988 and 1996 (0.1–5 kg ha^{-1} in a 5-km square) occur more frequently. These increases can be linked to areas with higher agricultural potential. Change in marginal areas such as large areas of the Highlands and Islands of Scotland, the Scottish Borders, Pennines, Welsh Hills and large urban conglomerations appear to have been much smaller.

Abatement scenarios

Abatement scenarios were modelled using the potential figures that might be expected under implementation of Integrated Pollution Prevention and Control (IPPC) in the UK (MAFF, 1997), which suggest anticipated reductions in NH_3 emissions as 4% for pigs and 7% for poultry. This equates to a total reduction of 2.9 kt for the UK. The effect of this would be a flattening of the very highest peaks of emission on the map. This would reduce the emission in the highest 1 km square by $\approx$160 kg ha^{-1} year^{-1} ($\approx$ a total reduction of 16 t in this square), although it should be noted that the 1-km estimates contain a very large uncertainty. In the highest 5-km square a reduction of 7 kg ha^{-1} year^{-1} is estimated ($\approx$ a total reduction of 17.5 t in this square). Because the pig and poultry sectors lead to the largest estimated emission density, these changes will focus reductions, and expected benefits, to the most acutely affected areas.

Uncertainties

The reliability of the overall magnitude of the emission estimates which are mapped here may to some extent be assessed by considering the closure of atmospheric budget estimates. This is addressed elsewhere by Fowler et al. (1998) and Sutton et al. (1998), and suggests that total ammonia emissions may have been underestimated or deposition overestimated. These uncertainties in the magnitude of total emissions do not, however, affect the procedures used here, which, if required, can be easily applied with different source strength estimates.

A number of points have to be raised regarding the spatial uncertainties in the results. These uncertainties can be divided into the following main categories:

- uncertainties /data quality of the model input data
- uncertainties introduced through the modelling process

Spatial uncertainties in the parish census are documented in Hotson (1986). One of the main problems here is that most of the time farm boundaries do not coincide with parish boundaries. Any farm may be counted with one parish for census purposes, but have the majority of its agricultural activity in one or several neighbouring parishes. This can not be circumvented as long as the spatial reference of the model input data is a parish membership. Checks can be carried out for crops by comparing areas of crops in the census data and potential locations for them on the landcover map. For grazing livestock (cattle, sheep, goats, deer and horses), stocking densities can be checked after redistribution within the parishes. In most cases this does not pose a too severe problem, although larger uncertainties may be expected for pig and poultry farms.

The landcover data pose another set of uncertainties, mainly related to unavoidable misclassifications occurring in the classification of satellite remotely sensed data. In addition, the spatial resolution of features on the ground compared to the spatial resolution of the sensor (30-m pixels) may pose a problem. This leads to an underestimation of landcover classes with smaller features and/or misclassifications due to a mixed pixel effect. This effect may be more noticeable in some areas of the country than others. A further uncertainty is introduced as landcover is not synonymous with landuse. Landuse had to be inferred from the landcover type identified through the satellite image classification.

As mentioned earlier, the NH_3 source strength estimates for livestock classes and crops applied in the present model are averages rather than spatially varying over the UK. In addition, the percentage contributions from the different parts of the livestock husbandry cycle such as housing, landspreading of wastes or grazing are averaged. This leads to underestimations in some regions and overestimations in others, depending on environmental conditions (such as climate, soil, land capability etc.) and farming practice in different areas of the country.

Further uncertainties are introduced through the modelling process. The rules for the redistribution of census items over the landcover data, e.g. stocking densities for grazing, are designed to match average farming practice, not taking any regional differences into account. At present, all emissions originating from one parish are redistributed into the same parish. In some instances, e.g. the spatial distribution of landspreading of wastes from large intensive livestock farming developments (pigs or poultry), this causes unrealistically large emissions within the boundaries of the concerned parishes, which abruptly stop at the parish boundaries. In reality, livestock manures from such developments are often spread over a much larger area. Further work is ongoing to remedy this.

Regarding the comparison between the 1988 and 1996 results, changes in the contents and spatial structure of the parish data also had to be taken into account. These changes included the introduction of new census items into later census questionnaires and differences caused by the availability of disclosive 1996 parish data for input into the model to provide greater detail (while ensuring non-disclosive model output).

Finally it is helpful to consider the spatial uncertainties within each 5-km grid area estimated by the present model. Since the 5-km estimates are mean NH_3 emissions, which are actually calculated at a 1 km level ($n = 25$), it is relatively straightforward to show other statistics for each 5-km grid element. An example of this is considered in Fig. 4, which shows the % coefficient of variation of the 1 km estimates (standard deviation/mean × 100). The interpretation of this map may not be immediately obvious, however, a closer inspection shows that the areas of high variability are primarily upland areas with an intimate mix of hills and intensively farmed valleys, and secondly intensively farmed areas with a mixture especially of pig and poultry emissions. The lowest variability is seen for emissions from cattle and sheep, where grazing land forms a major part of the landscape. Thus a high variability may be seen in the Lake District and Snowdonia (hills with intensively farmed valleys between) as well as in East Anglia (intensive pig and poultry emissions). The example of the Highlands of Scotland is interesting, as variability is high near the boundaries of agricultural areas, but much smaller in remote areas where there is little intensive agriculture even in valleys. Such a map as Fig. 4 includes both variability that is real due to local topography etc and variability due to model uncertainties. However, it illustrates the potential at a local level for 5-km maps

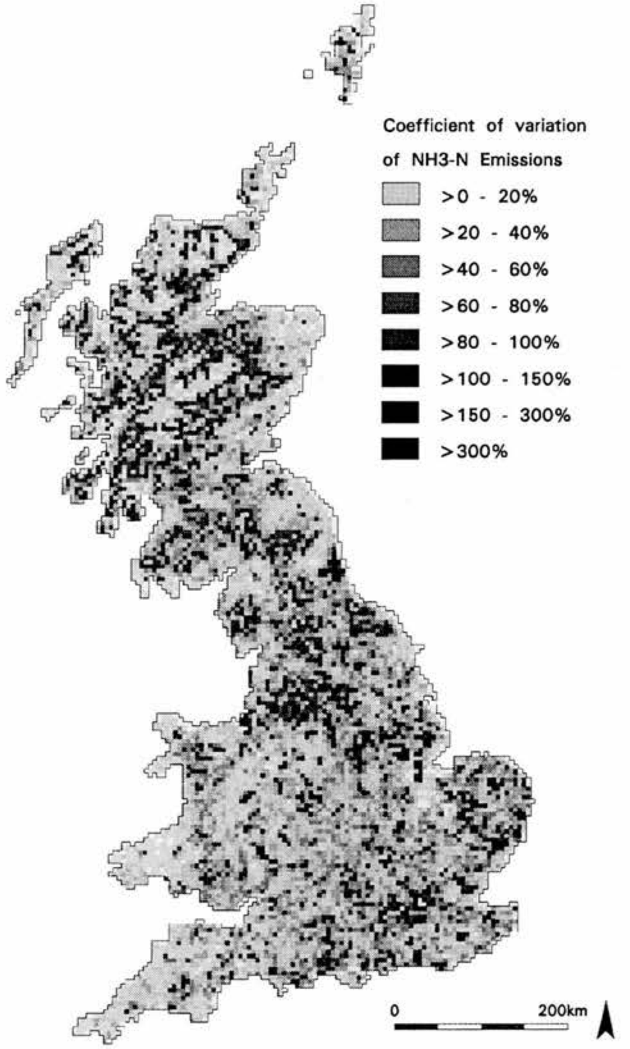

Fig. 4. Estimated variability in NH_3 emissions for Great Britain within 5-km grids expressed as the coefficient of variation (standard deviation/mean × 100).

hiding much of the variability. This will lead to both areas for which emissions are underestimated (intensive agricultural land) and areas where emissions are overestimated (hill areas, semi-natural land). Such features argue for the continued development of methods to improve the spatial resolution of NH_3 emissions estimates.

Conclusions

Ammonia emission maps have been developed for 1988 and 1996 modelled using a new methodology, which employs a specific spatial redistribution of census items as NH_3 emission sources (such as livestock housing and waste storage emissions, emissions from the landspreading of wastes, grazing emissions, etc.) onto suitable landcover types. The results of this are more realistic spatial NH_3 emission estimates than were possible with a previous more general redistribution. The new model is particularly an improvement for semi-natural/natural areas as it removes apparent emission sources from extensively used upland areas to the more intensively farmed lowland areas. This is especially important when NH_3 emission maps are used in conjunction with atmospheric transport, nitrogen deposition and critical loads exceedance models and to determine suitable abatement measurements in critical areas. At present the model results are mapped at a 5-km grid resolution to reduce uncertainty in the spatial location of NH_3 sources.

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THE SPATIAL DISTRIBUTION OF AMMONIA EMISSIONS IN GREAT BRITAIN FOR 1969 AND 1988 ASSESSED USING GIS TECHNIQUES

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Abstract - Maps of ammonia (NH_3) emissions are a key input for models describing the atmospheric transport and deposition of NH_3. An important question is whether emissions have changed over recent decades. There is evidence that total emissions have increased, largely since 1950, but up until now no attempt has been published to quantify the historical changes in spatial distribution of NH_3 emissions for Great Britain. The present paper shows how livestock numbers and crop areas have changed between 1969 and 1988. Using this information in 5 km grid form, together with information on N fertilizer use and livestock emissions, preliminary NH_3 emission maps have been constructed for 1969 and 1988. The results show substantial changes in the sources and patterns of NH_3 emission, related to changing agricultural policies and practice. In particular, there is an increased spatial variation in the emissions in 1988, probably due to larger farm sizes, however, there is currently significant uncertainty in the total magnitude of NH_3 emissions for 1969, due to differences in N input to livestock systems.

1. Introduction

The regional and national distribution of NH_3 emission estimates is of great interest for atmospheric transport modelling and assessing the effects of NH_3 deposition. In principle, measurements of NH_3 concentrations would be the best approach to quantify dry deposition to ecosystems, for comparison with maps of critical loads (e.g. INDITE 1994). However, the extreme spatial variability of NH_3 emissions and air concentrations would require a very large number of monitoring stations. On a national scale, gridded NH_3 emission inventories, together with application of atmospheric transport models, therefore provide essential tools to quantify the distribution of NH_3 deposition.

The spatial distribution of NH_3 emissions in the UK was estimated first by ApSimon *et al.* (1987) at a 10 km grid for England and Wales, and has been estimated more recently on a 5 km grid for GB by Eager (1992) and Sutton *et al.* (1995). The latter study provided NH_3 emissions for 1988, however effects of nitrogen deposition occur over decades, and the magnitude of past emissions and their distribution are also relevant. The present paper addresses the question of changes in the pattern of NH_3 emissions between 1969 and 1988. It integrates the agricultural emission mapping of Eager (1992) with emissions from non-agricultural sources and the most recent official emission factors (DOE 1995) on a 5 km grid for Great Britain.

Table 1: Total NH_3 emissions in the UK (1993) according to DOE(1995) and equivalent emission factors.

Source category	livestock (thousands)	total emissions (Gg NH_3 yr^{-1})	emission factor (kg NH_3 $animal^{-1}$ yr^{-1})
Cattle & calves	11729.0	160	13.64
Pigs	7753.8	30	3.87
Sheep & lambs	43901.0	20	0.46
Fowls	~~5453.0~~ 130,043	30	0.23
Tillage & cut grass	-	40	-
Non-agric. emissions	-	40	-
Total emissions	-	320	-

2. Methods

The main data source for spatially disaggregating NH_3 emissions in Britain is the June Agricultural Parish Census of MAFF. Using this information, agricultural emissions were scaled by animal numbers and crop areas, as summarized by MAFF (1973, 1990). These parish data were used in a 5 km grid format according to the reaggregation of Data Library, University of Edinburgh (Hotson 1988). Livestock emissions were scaled by emission factors per animal, and crop emissions derived information on average N application rates to crops (Boyd 1966, Church 1974, Dyer *et al.* 1989) together with an average fractional loss of applied N as NH_3.

Several estimates of NH_3 emissions in the UK have been provided recently (Sutton *et al.* 1995, Pain *et al.* 1995, pers. comm., ApSimon *et al.* 1995, pers. comm.). In the present study, equivalent NH_3 emission factors were derived from the official DOE figures (Table 1).

Non-agricultural sources were largely distributed by human population. The derived emission factors were used to scale the spatial distribution of NH_3 emissions according to 1969 and 1988 distribution. For this paper, the average 1993 emission factors for livestock (Table 1) were used unchanged for the 1969 and 1988 inventories, which is a signifcant source of uncertainty for the 1969 estimates. In contrast, the crop emissions for each year were calculated using data from the Survey of Fertilizer Practicefor the appropriate years (Boyd 1966, Church 1974, Dyer *et al.* 1989). Each of the components was implemented in the GIS system ARC/INFO 6.1 to provide the mapped distribution of emissions.

4. Results and discussion

The figures in Table 1 show that livestock agriculture represents by far the largest source of NH_3 in Great Britain. Hence changes in livestock numbers and distribution will be the main factors affecting the pattern of NH_3 emissions between 1969 and 1988. Changes in the total numbers and demographic structure of livestock between these years are shown in Table 2. The figures indicate that, with the exception of sheep, only relatively small changes in total numbers of livestock have occurred. However, there have been substantial changes in the demographic structure of each of the animal types. For example, there has been a decrease in the fraction of dairy cattle since 1969. This can be related the introduction of milk quotas (1984). Such changes in policy will have affected NH_3 emissions per animal because of the different N excretion from different animal sub-classes.

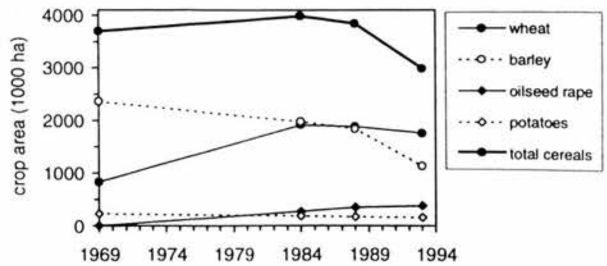

Figure 1: Changes in crop area in Great Britain between 1969 and 1993. Derived from MAFF (1973,1994).

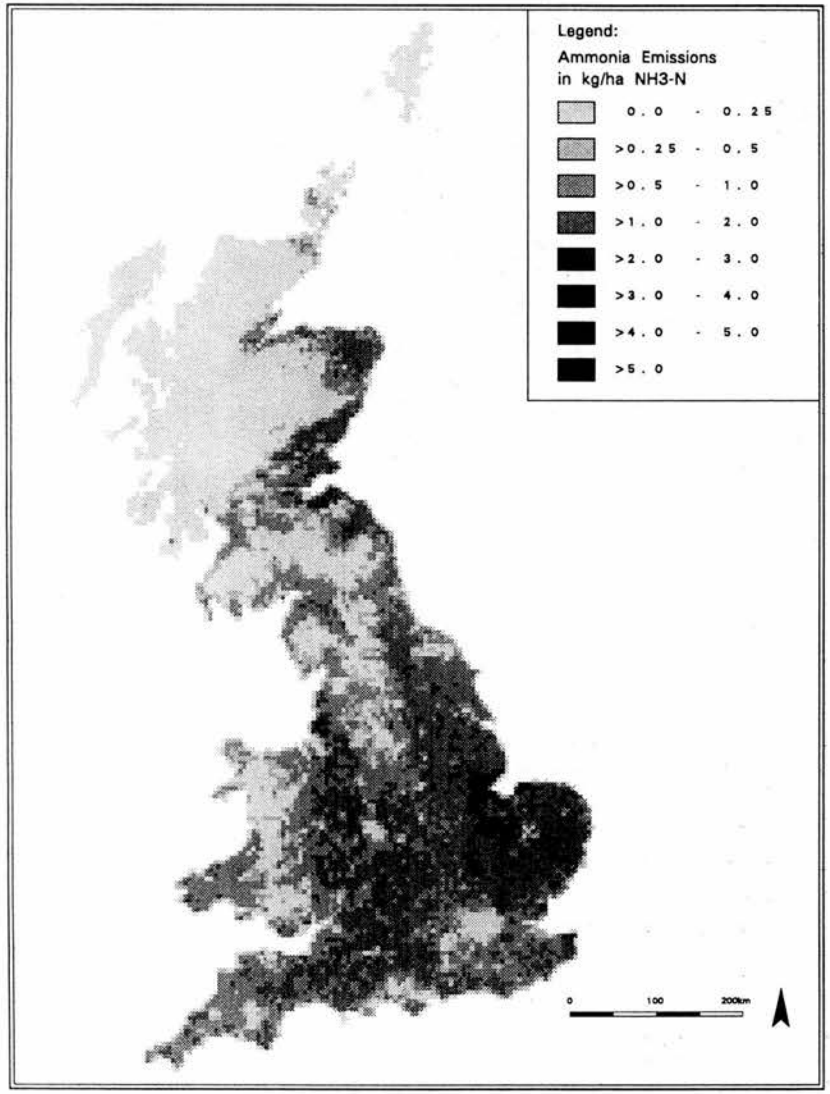

Figure 2. Ammonia emissions from fertiliser application and crops for 1969.

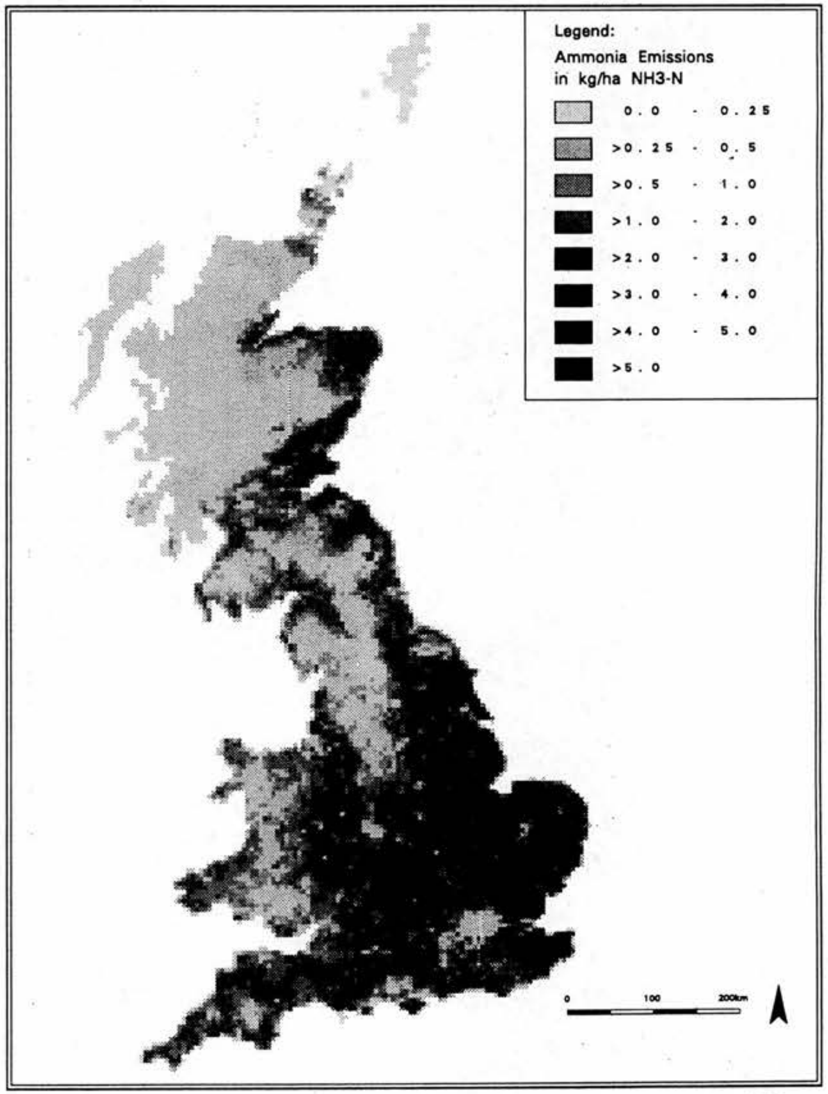

Figure 3. Ammonia emissions from fertiliser application and crops for 1988

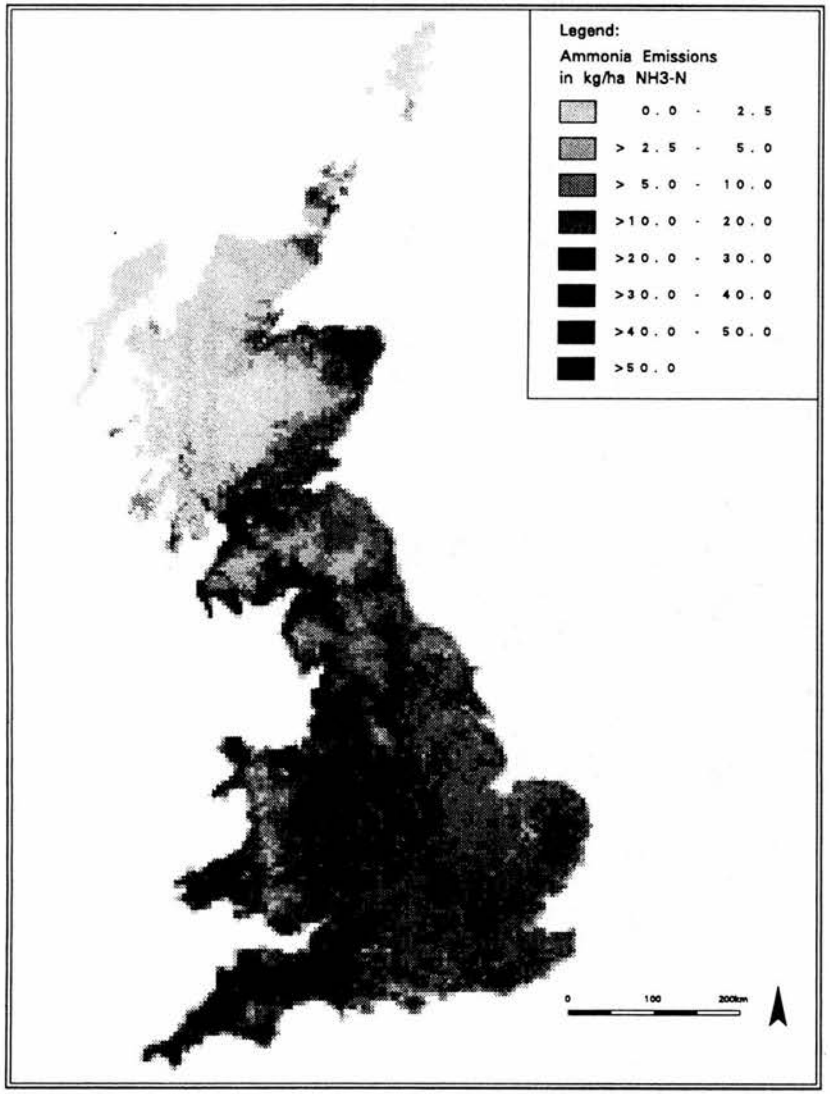

Figure 4. Total NH_3 emissions from agricultural and non-agricultural sources for 1969.

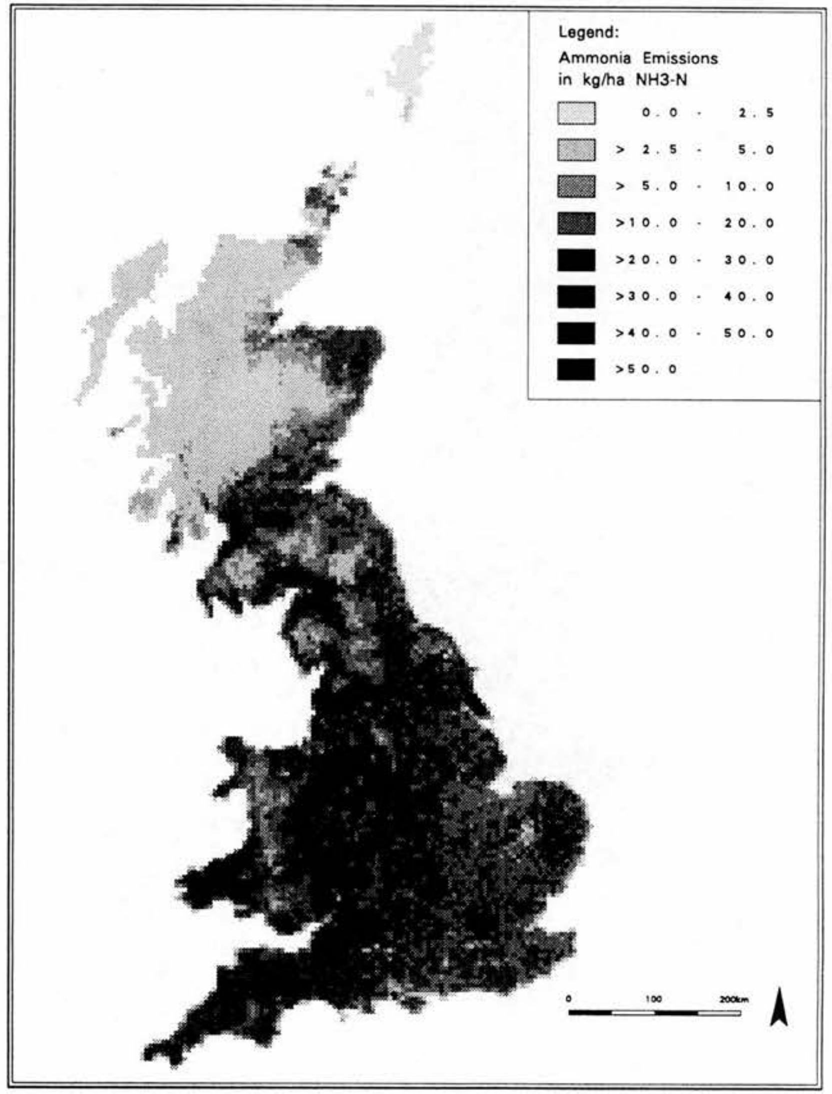

Figure 5. Total NH_3 emissions from agricultural and non-agricultural sources for 1988.

Table 2: Changes in livestock demography 1969-1988

Livestock	Livestock class	Animal numbers (1000s) 1969	1988	% change
Cattle	<2 years	2,234	2,747	23
	dairy cattle & calf cattle	4,685	3,662	-22
	bulls	85	48	-44
	fattening calves	1,597	1,627	2
	young fattening cattle	1,477	774	-48
	other cattle > 2 years	1,054	1,496	42
	total	11,132	10,355	-7
Pigs	breeding & other sows	818	805	-2
	boars	41	40	-2
	fattening pigs >50 kg	996	2,351	136
	fattening pigs 20-50 kg	3,048	2,086	-32
	piglets (<20kg)	1,844	2,042	11
	total	6,747	7,324	9
Sheep	ewes	13,711	18,274	33
	rams	327	783	139
	lambs	11,635	19,442	67
	total	25,673	38,512	50
Poultry (Fowls)	laying hens < 18 weeks	53,996	10,326	-81
	laying hens > 18 weeks	11,784	33,931	188
	all hens for breeding	5,626	5,607	0
	cocks & other table fowl	1,636	995	-39
	broilers	35,519	69,460	96
	total	108,561	120,319	11

Ammonia emissions and agricultural policy are closely linked. Asman *et al.* (1988) reported an 81% rise in NH_3 emissions across Europe between 1950 and 1980, which was estimated on the basis of changes in livestock numbers and fertilizer use. However, intensification is also expected to have increased NH_3 emissions per animal, particularly where these are fed grass with increased nitrogen content. These and other changes (changes in demography, animal breeding) make the identification of correct figures for historical livestock NH_3 emissions highly uncertain, and this is the subject of ongoing investigation.

It is clear that agricultural policy has had an increasing influence on farming in Britain. This trend was accentuated since the entry of the UK into the EC (1973), with the subsequent intervention of the Common Agricultural Policy (CAP). In addition to the effects on livestock, direct changes in crop production have also affected NH_3 emission. The areas of the major arable crops under cultivation in Britain over the period 1969-1993 are shown in Figure 1. This shows an increase in area under wheat and oilseed rape, both crops receiving high levels of N fertilizer. In addition, N inputs (kg N ha^{-1} $year^{-1}$) have generally increased over the period (Boyd 1966, Dyer *et al.* 1989). In contrast, the area of barley and of total cereals has decreased since 1988, as a result of CAP changes in response to overproduction of food, resulting in the introduction of 'set aside' (1988).

Examples of the mapped distribution of NH_3 emissions at 5 km x 5 km grid resolution for Great Britain are shown in Figures 2-5. The most dramatic changes have occurred for emissions from fertilizers and crops. Total crop emissions for 1988 are estimated at 43 Gg NH_3 $year^{-1}$ as compared with 32 Gg $year^{-1}$ in 1969, and this comparison is shown in mapped form in Figures 2-3. The change is a result of both changes in crop areas and increase in fertilizer application rate. Total ammonia emissions for 1988 and 1969 are estimated at 300 Gg $year^{-1}$ and 287 Gg $year^{-1}$, respectively. Although this is not a large difference overall, Figures 4 and 5 show how the spatial pattern of NH_3 emissions has changed substantially over the period. In 1988, there is a much larger spatial variability in emissions, probably associated with the increase of larger livestock farms as well as reduced activity on marginal land. This would be expected to have provided an increased impact of local NH_3 emissions in source areas.

It should be noted that the 1969 total emission map (Figure 4) is very largely affected by the contribution of NH_3 emissions from livestock. Given the changes in fertilizer application rates to grassland (and hence to animal feeds), there is considerable uncertainty over the emission factors to be applied, particularly for cattle and sheep. The present values were made using the same emission factors for different years, based on Table 1 for the most recent estimates. For example, although demographic changes would have favoured larger emissions in 1969 for cattle, smaller rates of fertilizer N input probably mean that emissions per animal were significantly smaller at this time. This aspect together with an assessment of the reliability of the spatial disaggregation are both areas requiring further study.

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